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Dissertation

Land use change and its effects on vegetation trends and fire patterns in Mediterranean rangelands

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Abstract

Drylands cover about 41% of the earth's surface, sustain the livelihoods of 2 billion people, harbor unique biodiversity and provide important ecosystem services, but land use and climate change exert considerable pressure on these ecosystems. However, pattern and drivers of land use change are weakly understood. Remote sensing can monitor these changes for large areas but methods to detect both gradual and abrupt events like fires are missing. The main objectives of this thesis were to develop remote sensing based methods to better quantify the impact of land use change on fire-prone Mediterranean ecosystems, and to apply these methods to better understand the influence of different land use processes on fire regimes. The island of Crete (Greece) served as study region where diverging land use transformation are extensive, fires are frequent and environmental gradients are large. First, the trade-off between different vegetation estimates when using Landsat-based trend analyses was quantified. The results suggested that simple vegetation estimates perform equally well and thus, allow for effective mapping of large areas. Second, a trajectory change detection approach was applied to separate gradual changes from abrupt events and to answer the question how land use systems and fire regimes have affected Crete's rangelands. Statistical modeling was then used to quantify the relative importance of land use processes in driving the fire regime. The results show that vegetation changes resulted in complex pattern of gradual changes and fires likewise. The fire regime appeared to be mainly driven by changing grazing systems. Fires were frequent in foothills whereas mountains showed increasing vegetation as a result of land abandonment. The statistical modeling confirmed that land extensification and climate are the primary drivers of fire regimes on Crete. The results suggest that the former fuel-limited fire regime will likely shift towards a drought-driven fire regime.

Zusammenfassung

Trockengebiete, die etwa 40% der globalen Landoberfläche abdecken, Lebensgrundlage für 2 Millionen Menschen bilden, einzigarte Biodiversität enthalten und wichtige Ökosystemdienstleistungen bereitstellen, sind beträchtlichem Druck durch Landnutzungs- und Klimawandel ausgesetzt. Die räumlichen Muster dieser Landnutzungsänderungen sowie deren Ursachen sind jedoch nur in Ansätzen verstanden. Fernerkundung kann Veränderungen großräumig beobachten, aber Methoden fehlen, um diese Veränderungen, die graduell und abrupt (z.B. Feuer) auftreten können, abzuleiten. Die Ziele dieser Dissertation waren die Entwicklung fernerkundlicher Methoden, um verschiedene Landnutzungsänderungen in mediterranen Ökosystemen zu quantifizieren und um den Einfluss verschiedener Landnutzungsprozesse auf das Feuerregime besser zu verstehen. Die griechische Insel Kreta wurde als Untersuchungsgebiet gewählt. Zuerst wurden basierend auf Trendanalysen von Landsatzeitreihen verschiedene Vegetationsmaße verglichen. Demnach führen einfache Vegetationsmaße zu ähnlich guten Ergebnissen und ermöglichen eine effiziente Beobachtung großer Gebiete. Anschließend wurden Veränderungstrajektorien abgeleitet, um graduelle und abrupte Prozesse zu unterscheiden und um zu analysieren, wie Landnutzungssysteme und Feuerregime sich auf Kreta ausgewirkt haben. Mittels einer statistischen Modellierung wurde der relative Einfluss von Landnutzungsprozessen auf das Feuerregime quantifiziert. Die Ergebnisse zeigten ein komplexes Muster von graduellen Vegetationsveränderungen und Feuern und deuten darauf hin, dass Feuer hauptsächlich vom Beweidungssystem abhängt. Feuer traten häufig am Fuße der Berge auf, wohingegen in den Berggebieten Vegetationszunahme vorherrscht. Die statistische Modellierung bestätigte, dass Extensivierung und Klima die Hauptursachen des kretischen Feuerregimes sind. So ist zu vermuten, dass sich das ehemals durch Brennmaterial limitierte Feuerregime zu einem trockenheitsgetriebenen entwickeln wird.

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Chapter I:
Introduction

1 Land use and global environmental change

Anthropogenic impact on the environment is manifold and has been accelerating dramatically over the past 50 years. These changes are now affecting the basic processes and functions of the earth system (MEA 2005a). For example, climate change triggered by anthropogenic emissions is accelerating (IPCC 2007), the majority of terrestrial ecosystems have already been altered or substantially transformed by human land use (Foley et al. 2005; Lambin and Geist 2006; Kareiva et al. 2007), no place on earth remains unaffected by anthropogenic activities of one form or another (Sanderson et al. 2002; MEA 2005c), and biodiversity is lost at unprecedented rates (Ehrlich and Pringle 2008; CBD 2010). Human domination of the earth system is now so profound that anthropogenic biomes (i.e., anthromes) have been suggested (Ellis and Ramankutty 2008). Humans are today changing the earth more rapidly and extensively than in any other period and some argue therefore that an entirely new era in planetary history, the Anthropocene, has arisen, in which human actions have become the main driver of global environmental change (Crutzen 2002).

As global change progresses, there is also an increasing awareness that humanity entirely depends on ecosystems and the services they provide (MEA 2005c). These services encompass provisioning services (e.g., food, fresh water, wood and fiber), supporting services (e.g., nutrient cycling, soil formation), regulating services (e.g., climate stability flood protection, disease regulation), and cultural services (e.g., spiritual value, recreational opportunities). As human activities have altered the distribution of and functioning of ecosystems, many of these services have been substantially degraded or lost altogether in some regions (MEA 2005c; Perrings et al. 2010; Raudsepp-Hearne et al. 2010; Isbell et al. 2011). Indeed, regarding a range of parameters humanity likely has overstepped planetary boundaries and is currently outside the safe operating space (Rockstrom et al. 2009).

Land use change is the primary driver of global environmental change, either by converting natural landscapes for anthropogenic use or by changing management practices on human-dominated lands (GLP 2005). Land use is essential for human well-being, and for meeting humanity's demands for food, and bioenergy (MEA 2005c). Farming has been the main driver of population growth and of the success of the human enterprise in general (Boserup 1964; Diamond 2005; Foley et al. 2011). As a result, agriculture is by far the most widespread land use covering about 35% of the earth's ice-free land, (Ramankutty and Foley 1999)). Agricultural production has been increasing exponentially throughout the last two centuries, both regarding expansion of agricultural area (Ellis et al. 2010; Klein Goldewijk et al. 2010) and in terms of agricultural intensification with yields outpacing global human population growth (Matson et al. 1997; Foley et al. 2011).

Unfortunately, the expansion of food, fiber and bioenergy production has substantial trade-offs. Land use is the single largest source of emissions of greenhouse gases via converting carbon-dense forests to croplands and via emissions related to particular land use practices, e.g., rice fields, livestock breeding, and artificial fertilizer use (IPCC 2007; Leadley et al. 2010; Foley et al. 2011). The current biodiversity crisis is mainly driven by land use change, via the destruction, degradation, and fragmentation of habitat (Fischer and Lindenmayer 2007; Koh and Ghazoul 2010). Land use also results in a massive fertilizer input, leading to deterioration of freshwater quality and coastal areas (Potter et al. 2010; Bouwman et al. 2011). These are only some of the negative impacts of land use on the earth system, but they illustrate that the global land use system is currently far from sustainable.

Land use will have to expand further to satisfy humanities growing demand for resources (Foley et al. 2011). Human population is expected to rise to 10 billion people by the end of the 21st century (Lee 2011), consumption is increasing further as more and more people shifts towards diets richer in meat (Erb et al. 2009), and bold bioenergy goals are being discussed (WBGU 2009; Lotze-Campen et al. 2010). One of the grandest challenges that humanity currently faces is to substantially expand land use in the coming decades while transforming land use to sustainability (Foley et al. 2011; Tilman et al. 2011). Primarily, this requires a better understanding of where land use changes and what drives these changes.

2 Drylands and Mediterranean ecosystems

About 41% of Earth's land surface is covered by dryland ecosystems. Drylands occur on all continents between 63° N and 55° S latitude where plant production is limited by water availability, and are defined as ecosystems where the ratio of total annual precipitation (P) to potential evapotranspiration (ET) is less than 0.65 (Middleton and Thomas 1997). The aridity index (AI=P/ET) (AI) captures this ratio and is used to further subdivide drylands into four categories: dry sub-humid (AI = 0.50–0.65), semi-arid (AI = 0.20–0.50), arid (AI = 0.05– 0.20), and hyperarid (AI < 0.05) regions (Fig. I-1). Drylands support a large number of ecosystems which are a part of four broad biomes: forest, Mediterranean, grassland and desert (Safriel et al. 2005).

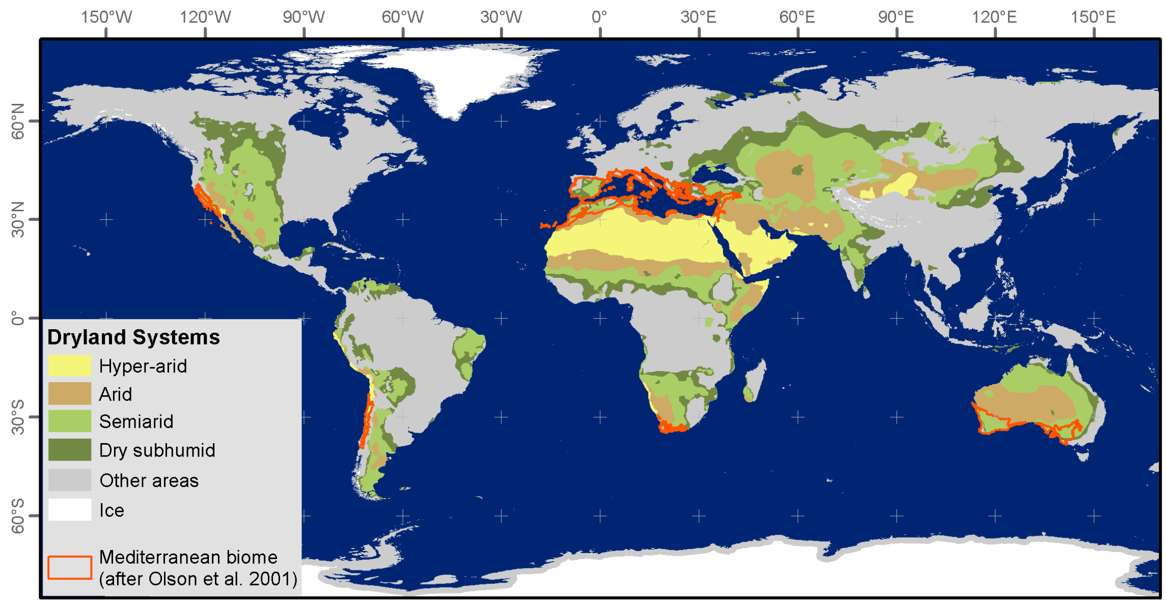

Figure I-1: Dryland systems (data source: UNEP - http://geodata.grid.unep.ch/) and Mediterranean-type ecosystems (data source: WWF - http://www.worldwildlife.org/science/data/item1875.html).

More than 2 billion people (about one third of the world population) inhabit drylands. At least 90% of these people dwell in developing countries and their livelihoods directly depend on dryland ecosystem services including the provision of food, forage, water resources and wood fuel (MEA 2005b). Drylands also provide ecosystem services of global significance. For example, dryland soils sequester and store vast amounts of carbon, due to the large areal extent, thereby contributing substantially to climate stability (Lal 2004). Drylands are also of outstanding importance in terms of their conservation value, for example harboring about half of the world's endemic bird areas, one fourth of all protected areas in the IUCN system, and one sixth of all global centers of plant diversity (White and Nackoney 2004).

Drylands are typically characterized by harsh environmental conditions, which include generally little and highly variable rainfall, high temperatures, high evaporation, and cyclic droughts. Moreover, dryland soils are characterized by comparatively low organic matter content, low nutrient reserves, and low nitrogen content (Skujins 1991). Vegetation has evolved to survive under these conditions and developed a variety of morphological, physical and chemical adaptations, ranging from drought-enduring (i.e. xerophytes) to drought-avoiding (i.e. ephemeral annual grasses) traits. Plant adaptations show typical features like xeromorphological leaf structures, physiological controls of transpiration and metabolism rates, moisture and nutrient storage organs, and thorns (Hassan and Dregne 1997). These adaptations become less pronounced with decreasing aridity as the conditions for plant growth become more favorable (Mittermeier et al. 2005).

Among the dryland ecosystems, Mediterranean-type ecosystems cover a relatively small area (4% of all drylands) and occur within semi-arid and dry subhumid areas (Lal 2004). Mediterranean ecosystems are defined via their climate characteristics with hot, dry summers and cold, wet winters and include non-dryland areas. Mediterranean ecosystems occur between 30 and 40 degrees latitude in five disjunct regions around the globe (Fig I-1): the Mediterranean Basin, California, Chile, South Africa, and southwestern and southern Australia (Vogiatzakis et al. 2006).

Mediterranean ecosystems are outstanding in terms of their biodiversity, containing for example 48,000 plant species which equals 20% of the world's vascular plant species richness (Cowling et al. 1996). Many of these species are exceptionally rare and endemic to the Mediterranean. As a consequence, all Mediterranean regions are hotspots of global conservation concern (Myers et al. 2000). This is particularly true for the Mediterranean Basin, the largest of all Mediterranean regions, which harbors exceptionally biodiversity due to its location at the intersection of two major landmasses, Eurasia and Africa (Mittermeier et al. 2005). The Mediterranean Basin itself hosts half of the vascular plant species, but biodiversity varies widely and 44% of the entire endemism is found in 10 regional biodiversity hotspots covering 22% of the entire Mediterranean Basin (Medail and Quezel 1997).

Although Mediterranean ecosystems differ greatly in terms of geology, soils, floristic and faunistic compositions, there are considerable similarities in ecosystem structure and dynamics (Vogiatzakis et al. 2006) attesting of convergent evolution (DiCastri, 1981). Mediterranean vegetation (Fig II-1) is characterized by woody shrubs with sclerophyllous leaves which have a variety of local names, namely, maquis and garrigue in the Mediterranean Basin, matorral in Chile and Spain, chaparral in California, macchia in Italy, mallee in Australia and renosterveld in South Africa (di Castri 1981).

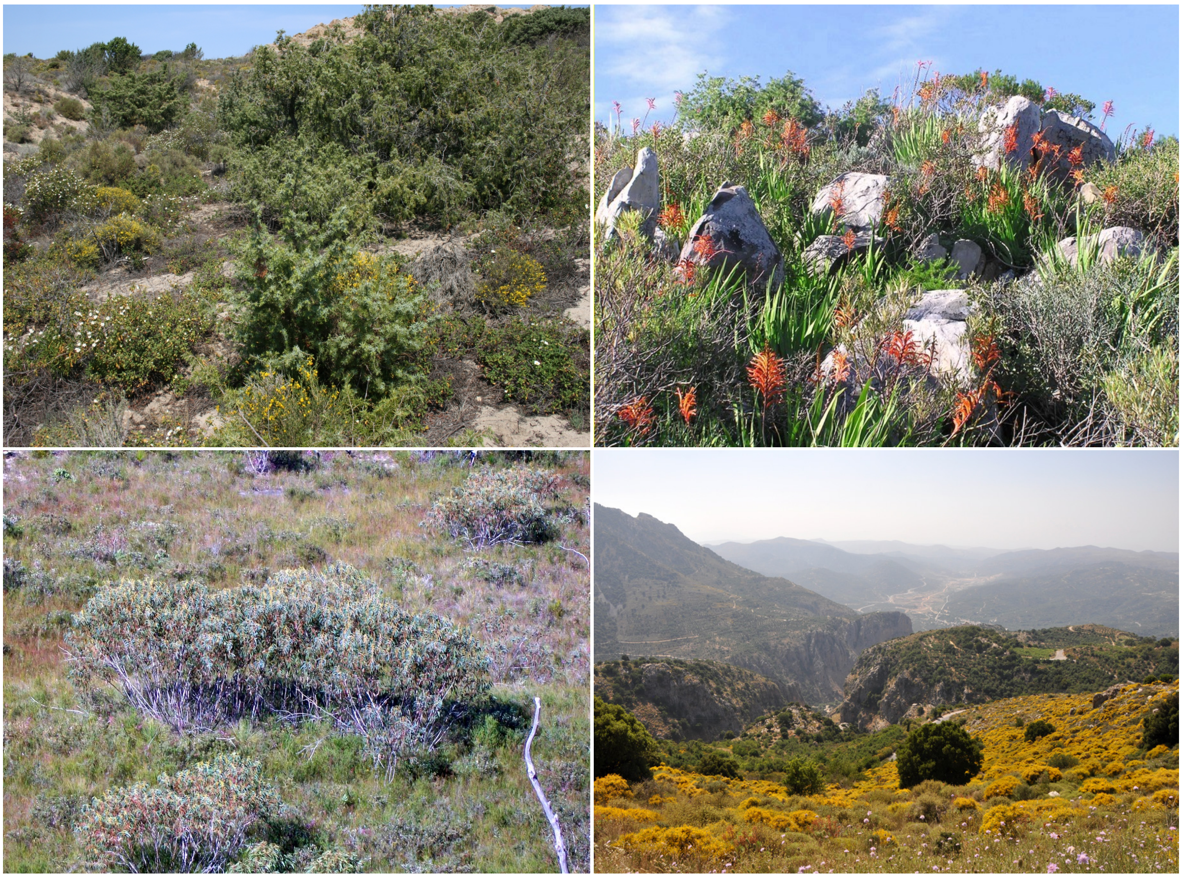

Figure I-2: Typical Mediterranean-type landscapes - above left: Garrigue, Corsica, above right: renosterveld, South Africa, below left: mallee, Australia, below right: Crete). Photos downloaded from commons.wikimedia.org.

Fire is a natural disturbance agent in all dryland ecosystems that contain vegetation cover, and especially in Mediterranean ecosystems which are fire prone due to the seasonal drought during the summer months that results in a high fire risk and high flammability of the vegetation. Fires have been incorporated in the evolution of Mediterranean ecosystems for Millenia shaping their diversity (Cowling et al. 1996) and their structure and function (Bond and Keeley 2005). Vegetation adaptations to fires have evolved independently in all Mediterranean ecosystems and convergent fire-related plant traits are found on different continents. These range from adaptive and protective mechanisms that allow plants to survive fires (e.g. the thick, insulating bark of *quercus suber*) to specialized postfire persistent traits which include resprouting from seed banks, serotiny, and fire stimulated germination by heat and smoke (Keeley et al. 2011).

3 Land use in drylands and Mediterranean ecosystems

Drylands have long been utilized by humans, including for a variety of land uses. The primary land uses in drylands are livestock grazing and crop cultivation. The vast majority of drylands that can support vegetation are used as rangelands (69%) which sustain about 50% of the world's total livestock population. Croplands cover approximately 25% of all dryland areas (Reid et al. 2004). The major constraint to land use in drylands is the low and highly variable rainfall. This variability makes drylands highly vulnerable to disturbances, including land use, and many examples exist where past land use has resulted in the degradation of drylands (e.g., in Chile, the Mediterranean Basin, or the Sahel, an elaborate compilation of land degradation related studies can be found for instance in Geist (2005)). Yet, where land management systems have evolved over Millennia, land use has also often adapted to the conditions of drylands, and traditional land use systems are therefore also often characterized by high resilience. Livestock systems involve nomadic and transhumance movements and are often not specialized but paired with farming, including field crops (Hassan and Dregne 1997). However, land use varies largely among dryland climates and the proportion of livestock increases with aridity from 34% in subhumid regions to 97% in the hyperarid region (MEA 2005b). Likewise, arable cultivation is restricted to semi-arid and dry sub-humid regions or where irrigation is possible.

In terms of land use, the Mediterranean biome is one of the most transformed biomes on earth (MEA 2005c). This is especially evident in the Mediterranean Basin, where only five percent of the original extent of the ecoregion contains relatively natural vegetation, placing the Mediterranean Basin among the four most drastically altered hotspots on Earth (di Castri 1981). The Mediterranean Basin is also the region with the longest land use history, as the region partly overlaps with the fertile crescent (one of the regions where agriculture was invented) and human settlement and farming already began 8000 years ago (Diamond 2005; Bramanti et al. 2009).

Likewise, the Mediterranean Basin has the longest history of human fire use among all Mediterranean ecosystems. Fire has for very long been an important land management tool in the Mediterranean Basin, and used by pastoralists to improve rangeland conditions and by farmers to clear new land for cultivation (Naveh 1975a). The long and intensive human impact by various activities such as fire, grazing and cutting shaped the characteristic small-scaled mosaics of different land uses in Mediterranean landscapes, resulting in today's typical Mediterranean cultural landscapes (Le Houérou 1981). Smallscale farming, livestock husbandry and agropastoralism have long been the primary land uses with characteristic adaptations like terracing and transhumance practices.

Land use differs substantially within the Mediterranean Basin, and especially among the northern and southern rim of the Mediterranean Sea. In the European part of the Mediterranean Basin, widespread socio-economic changes have occurred since mid of the 20th century. The Mediterranean Basin has also experienced a boom in tourism, now being Europe's prime tourist destination, which has led to the large-scale transformations of many coastal regions for touristic development. Finally, as elsewhere in Europe, rural populations have declining substantially and strong urbanization trends have been widespread resulting in urban development and urban sprawl (Le Houerou 1993). The region became part of the European Union, resulting in a fundamental restructuring of the regions agricultural sectors. Most importantly, subsidy payments under the Common Agricultural Policy (CAP) became one of the main drivers of land use decisions in these countries (Lorent et al. 2008; Lorent et al. 2009). Together these socio-economic trends have resulted in a polarization of land use with an intensification of land use in productive areas (e.g., expansion of irrigated areas and greenhouses) and the abandonment of marginal areas and traditional land use systems. Where and how these land use changes occur, however, remains weakly understood, mainly because these processes often co-occur in the same region, because land use patterns are fine-scaled and often highly heterogeneous, and because the drivers of intensification and extensification are not fully understood (Bernues et al. 2005; Tzanopoulos and Vogiatzakis 2011).

These land use changes have altered a range of ecosystem services. For instance, substantial concern has been expressed about land degradation due to land use concentration (e.g., via overgrazing (Stafford Smith et al. 2009)). Land use intensification such as the expansion of irrigation facilities has also resulted lower ground water tables and fresh water shortage in some areas. Likewise, land use changes in the Mediterranean Basin have impacted the region's biodiversity. Intensification leads to a fragmentation and loss of habitat (Peña et al. 2007). On the other hand, many Mediterranean species are inherently connected to the traditional, low-intensity land use systems that have evolved over Millennia. The ongoing polarisation of land use leads to a loss of these cultural landscapes and with them biodiversity is lost, too (Sala et al. 2000).

Furthermore, the diverging land use trends of intensification and extensification have also implications for the fire regimes in the Mediterranean Basin. Fire is a traditional land management tool that has been widely used by shepherds for shaping rangeland vegetation and for improving fodder quality and availability. Both intensification and extensification have been suggested to affect fire regimes. For instance, land use intensification may lead to a concentration of grazing and shepherds and thus intentional ignitions (Papanastasis 2004). On the other hand, rural exodus and the abandonment of traditional land use practices such as transhumance livestock husbandry may result in fuel accumulated and increasing fire risk (Viedma et al. 2006; Duguy et al. 2007; Röder et al. 2008a). Fires increasingly put human infrastructure and lives at risk in the Mediterranean Basin, and understanding the importance of land use relative to other drivers of fire regimes (e.g., climate) and how different land use changes are linked to fire regimes is therefore of great importance.

4 Mapping land use change in drylands

Understanding land use changes and how they impact the environment, including via altered fire regimes, first and foremost requires an understanding of *where* land use changes. Remote sensing is arguably the most important and most valuable tool at hand for this task via mapping changes in land cover. A vast variety of sensors have been launched in the last decades covering different temporal, spatial and spectral scales which allow the monitoring of ecosystems from local to global scales, over long time periods, and in a consistent manner. Land use changes result either in land cover conversions, i.e. the entire replacement of one land cover type by another (e.g., agricultural expansion, urbanization) or in land cover modifications, i.e. the change of land cover attributes within a given land cover class (e.g., vegetation change due to changes in grazing pressure) (Lambin and Geist 2006). Land change research so far has focused predominantly on the detection of land use conversions, although gradual changes may be as widespread as conversions. Only since a few years, there is an increasing awareness of the importance of land cover modifications associated with gradual changes in land management.

The detection of gradual land cover modifications principally relies on the detection of trends in land cover's biophysical attributes. Vegetation cover is the most suitable biophysical attribute for characterizing ecosystem conditions and dynamics, due to its distinct spectral signature. However, the detection of subtle changes in the vegetation signal induced by land use change is challenging. Drylands and Mediterranean ecosystems exhibit high seasonal variability in vegetation vigor and cover, which are driven by the high variability in temperature and rainfall, as well as abrupt changes caused by disturbances like fires. Although a large variety of change detection techniques have been developed to capture land change by comparing pairs of images (e.g., image differencing, change vector analysis, composite analysis, etc, (Coppin et al. 2004; Lu et al. 2004)), these methods often fail to detect subtle changes in land use and land cover. The detection of gradual land modifications requires image time series to separate natural variability from long-term changes in vegetation due to land use and in order to identify and separate long-term trends and disturbance-type changes by continuous-variable approaches.

Trend analyses have frequently been applied at broad scales using remote sensing imagery from coarse-scale sensors such as NOAA AVHRR or MODIS, including for assessing dryland ecosystems (Wessels et al. 2007; Hill et al. 2008; Seaquist et al. 2009). However, the time period covered by such sensors is often short (e.g., 10 years in the case of MODIS), making the detection of transient vegetation changes challenging. The spatial resolution of these sensors is often also too coarse to reliably resolve small-scaled structures that are typical for Mediterranean-type, heterogeneous landscapes (Stellmes et al. 2010).

The opening of the United States Geological Service (USGS) Landsat archives in 2009, and the subsequently changing data access policy of several other archives, include the ESA Landsat archive, has enabled new and exciting opportunities (Cohen and Goward 2004; Wulder et al. 2011). Landsat satellites have collected the longest running time series of remote sensing data, resulting in a truly unique data record. The Landsat program consists of a series of seven sensors that have operated successively since the 1970s, when the first of three Landsat Multispectral Sensors (MSS) was launched. Since Landsat Thematic Mapper (TM) 4 was brought into orbit in 1982, consistent imagery at a spatial resolution of 30 x 30 m and a spectral resolution of seven bands (including bands in the near and middle infrared), have been collected. The fine spatial resolution enables land cover and land use change mapping at the scales commonly associated with human land management, and the spectral resolution covers all wavelengths important for detecting ecosystems processes (Cohen and Goward 2004). Today, the Landsat 7 ETM+ mission, the subsequent operation after the TM 4 and 5 satellites, is the only currently operating system but has a scan line malfunction since 2003 which results in a loss of about 20-25% of every full scene. Landsat TM 5 was also still functioning at the time of writing of this thesis although image acquisition stopped by the end of November 2011 due to a degrading electronic component (www.usgs.gov). In recognition of the long and unique data record and the importance of Landsat sensors for land change science, the Landsat Data Continuity Mission (LDCM) has high priority in NASA mission planning and is currently scheduled for 2013.

Many studies have sought to exploit the temporal depth of the Landsat archives, even before the opening of the archives. These studies captured disturbances events and characterized subsequent vegetation recovery, like volcanic eruptions (Lawrence and Ripple 1999) or fires (Viedma et al. 1997; Röder et al. 2008a). With the opening of the USGS archives, there was an increasing interest in developing approaches to detect gradual and disturbance-type changes at the same time (Kennedy et al. 2007; Huang et al. 2010; Kennedy et al. 2010). Much attention has been paid to boreal and temperate forest ecosystems in that context (Huang et al. 2009; Griffiths et al. 2011; Schroeder et al. 2011). However, analyses of Landsat time series for drylands and Mediterranean-type ecosystems are sparse, and most existing studies have focused on Australian ecosystems, where large-area monitoring programs based on Landsat imagery have been implemented (Wallace et al. 2004; Wallace et al. 2006).

Effectively mapping vegetation dynamics due to land use change in drylands and Mediterranean ecosystems, requires addressing two questions. First, the choice of vegetation estimate in trend analyses remains weekly understood. This is especially important in dryland environments where vegetation cover is low and the vegetation signal affected by soil and bedrock signals. Quantitative methods such as spectral mixture analysis (SMA), which quantifies the abundance of vegetation within a single pixel, have often been suggested to outperform simpler methods (e.g., vegetation indices) when assessing vegetation properties in single-date studies (Elmore et al. 2000). However, applying SMA across large areas can be challenging, which is a strong limitation in dryland monitoring. The trade-offs of using simpler methods, which in turn can be more easily and robustly be implemented across large areas, have so far not been quantified.

Second, all trajectory change detection approaches that explore the depth of the Landsat archive have been developed for and exclusively applied to temperate forest ecosystems, where the vegetation signal is high and yearly variations in phenology are small. Yet, their usefulness for mapping vegetation change in drylands and Mediterranean ecosystems, where subtle vegetation changes and abrupt disturbances (e.g., via fire) co-occur, where vegetation communities are complex and contain sparse canopy cover, and where strong inter-annual variations in phenology prevail, remains untested. Solving these challenges will enable an improved mapping of the rates and patterns of land use and land cover dynamics in drylands and Mediterranean ecosystems, thereby providing a much-needed basis for better understanding the drivers and impacts of land use change on the regions ecosystems.

5 Research questions and structure of the thesis

This thesis sought to develop methods to better quantify the impact of land use change on Mediterranean ecosystems across large areas, and to apply these methods to better understand the influence of different land use processes on fire regimes. The island of Crete is the largest island of Greece and the fifth largest in the Mediterranean was chosen as the study region for this thesis. Crete, as many areas in the Mediterranean, has experienced marked socio-economic changes during the last decades, including the massive urbanization, tourism facilities expansion, and land use polarization trends discussed above. The island is also characterized by strong environmental gradients, and fires are frequent. Together, this makes Crete an ideal case study for developing remote-sensing based methods to quantify changes in land use and land cover, and for addressing the link between land use and fire regimes.

This thesis asked three specific research questions:

1.) What are the trade-offs between different vegetation estimates when using trend analyses to assess gradual vegetation change in Mediterranean ecosystems?

2.) How have different land use change processes and fires reshaped vegetation communities on Crete?

3.) What is the link between different land use processes and fire regimes on Crete?

This thesis contains five chapters. The introduction is followed by three main chapters, Chapters II, III, and IV, which were written as stand-alone manuscripts to be published in international peer-reviewed journals. Each of these main chapters is therefore structured into the subsections introduction, study region, methods, results and discussion, thereby resulting in a limited amount of recurring material throughout the thesis. These three chapters were prepared, submitted, and published as follows:

Chapter II: Sonnenschein, R., Kuemmerle, T., Udelhoven, T., Stellmes, M., and Hostert, P. (2011): Differences in Landsat-based trend analyses in drylands due to the choice of vegetation estimate, *Remote Sensing of Environment*, 115:1408-1420

Chapter III: Sonnenschein, R., Kuemmerle, T., Kennedy, R.E., Cohen, W.B., Hostert, P., (2011): Analyzing dense Landsat time series to assess the relationship between fire and grazing on Crete (Greece), *Remote Sensing of Environment*, submitted

Chapter IV: Sonnenschein, R., Kuemmerle, T., Baumann, M. (2011): Analyzing the relative importance of climate, people, and land use in driving fire patterns on Crete (Greece), *Regional Environmental Change*, in preparation

Following the three main chapters, the dissertation concludes with a synthesis and outlook chapter.

Chapter II:
Differences in Landsat-based trend analyses due to the choice of vegetation estimate

Remote Sensing of Environment 115(2011): *1408-1420*

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Abstract

Drylands cover about 41% of the globe's surface and provide important ecosystem services, but land use and climate change exert considerable pressure on these ecosystems. Both of these drivers frequently result in gradual vegetation change and landscape-scale trend analysis based on yearly vegetation estimates can capture such changes. Such trend analyses based on high-resolution time series of satellite imagery have so far not widely been used and existing studies in drylands relied on different vegetation measures. Spectral mixture analysis (SMA) has been chosen due to its superiority to simpler vegetation estimates in quantifying vegetation cover in single-date studies, however SMA can be challenging to implement for large areas. Here, we quantify the trade-off involved when using simple vegetation estimates instead of SMA fractions for subsequent trend analyses. We calculated NDVI, SAVI and Tasseled Cap Greenness, as well as SMA green vegetation fractions for a time series of Landsat images from 1984-2005 for a study region in Crete. Linear trend analysis showed that trend coefficients and the spatial patterns of trends were similar across all vegetation estimates and the entire study region, especially for areas where vegetation changed gradually. On average, trends based on simple measures differed less than 5% from SMA-based trends with decreasing similarity in trend results from Tasseled Cap Greenness to SAVI and NDVI. Vegetation estimates differed markedly in their response to disturbance events such as fires. Trend analyses based on qualitative measures can easily be applied across very large areas and using multi-sensor time series based on high-resolution data. While the subtle differences between vegetation estimates may still be important for some applications, the robustness of trend analyses regarding the choice of vegetation estimate bears considerable promise to reconstruct fine-scale vegetation dynamics and land use histories and to assess climate change impacts on the world's drylands.

1 Introduction

Changes in land use are among the key drivers of global environmental change and remote sensing is the most important tool for monitoring land use effects on ecosystems. Though many change detection methods have been developed (Coppin et al. 2004), the question of how to map land use change reliably remains a central challenge (Turner et al. 2007; Pausas and Fernández-Munoz 2010). This is partly because land use change can result in both land cover conversions (i.e., changes from one land cover to another) and land cover modifications (i.e., subtle changes within a land cover class), and remote sensing largely focused on mapping the prior. However, land cover modifications may be more prevalent and significant than conversions (Donald et al. 2002; Huang et al. 2002). For example, forest degradation in the Amazon may be more widespread than deforestation (Foley et al. 2007), agricultural intensification is a main driver of biodiversity loss (Tilman et al. 2001), and overgrazing results in rangeland degradation (Asner et al. 2004; Stafford Smith et al. 2007).

There is a strong need for methods that can map such subtle land cover changes (Turner et al. 2007). The problem is that land cover modifications are often obscured by differences in illumination, atmospheric conditions, or phenology between images, especially outside the tropics, where seasonal variability is high. Separating true land cover change from pseudo-change and phenology using traditional approaches based on image pairs is often challenging (Coppin et al. 2004). Time series analyses provide powerful alternatives (Udelhoven 2006), because seasonal variations can be separated from long-term trends. Such trend analyses have frequently been applied to characterize land surface phenology change from coarse-scale imagery (Slayback et al. 2003; Heumann et al. 2007; Bradley and Mustard 2008). Yet, the spatial resolution of these sensors is often too coarse to detect subtle land cover changes, especially in fragmented, human-dominated landscapes that comprise the majority of the world's terrestrial ecosystems (Ellis and Ramankutty 2008). Moreover, most coarse-resolution sensors have relatively short data records, thereby inhibiting trend analyses over time-scales in which land cover modifications manifest.

The unique, long-term data record of Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper (ETM+) sensors offers great possibilities to fill this gap. The Landsat archives constitute the world's longest time series of high-resolution earth observation imagery and have been the primary data source for ecosystem monitoring. Yet, only recently researchers have begun to make full use of the temporal depth of these archives (Lawrence and Ripple 1999; Hostert et al. 2003a; Kennedy et al. 2007; Röder et al. 2008b; Huang et al. 2009; Kennedy et al. 2010).

These studies suggest that trend analyses of multi-year Landsat image time series has tremendous opportunities to advance our understanding of the rates and spatial patterns of land cover modifications, because trend analyses separate year-to-year dynamics from gradual land cover trends even if these trends are subtle. This offers potential for overcoming some of the limitations of traditional change detection methods and enables mapping for regions where land use practices dominantly result in gradual land cover change. However, so far Landsat trend analyses have mostly been used to characterize abrupt land cover changes such as volcanic eruptions (Lawrence and Ripple 1999), fire events (Viedma et al. 1997; Röder et al. 2008a), or forest disturbances (Kennedy et al. 2007).

Drylands are a prime example of an ecosystem where land use often results in gradual land cover change (Geist and Lambin 2004; MEA 2005b). Drylands, encompassing the sub-humid, semi-arid, arid, and hyper-arid climates, cover about 41% of the Earth's land surface and are inhabited by more than 2 billion people whose livelihoods directly depend more on dryland ecosystem services than elsewhere (MEA 2005c). These ecosystems comprise one quarter of the global soil carbon reserve (MEA 2005b) and they are rich in biodiversity. Drylands are characterized by harsh environmental conditions, including scarce and highly variable rainfall and shallow soils (Reynolds et al. 2007). Land use affects dryland ecosystems mainly via land cover modifications such as vegetation loss due to overgrazing, woody encroachment due to agricultural abandonment, especially in drylands with millennia of land use such as the Mediterranean Basin. Drylands are also characterized by tipping points in ecosystem response to disturbance, potentially triggering degradation (Rietkerk et al. 2004), and monitoring gradual changes in these systems is therefore important (Reynolds et al. 2007).

Trend analysis has been routinely applied to Landsat time series to monitor the wide pasture land in Australia (Wallace et al. 2004; Wallace et al. 2006). However, only a limited number of studies used dense Landsat TM/ETM+ time series to map land use modifications in other dryland environments, i.e., the impact of grazing pressure has been characterized in Utah (Washington-Allen et al. 2006) and Nepal (Paudel and Andersen 2010). Overgrazing and decreasing grazing pressure between 1984-2000 were also identified for a small study region in Northern Greece (Röder et al. 2008b). Similarly, gradual vegetation change due to grazing occurred on Central Crete, Greece (Hostert et al. 2003a). Trend analysis also helped to separate vegetation change due to climate variability from abrupt human-induced disturbances (e.g., fires) (Röder et al. 2008a). These studies highlight the potential of Landsat-based trend analyses to effectively map land cover modifications in drylands. Overall though, trend analysis has not been used extensively for dryland monitoring and existing examples focused on small study region with the exception of Australia's rangeland monitoring programs (Wallace et al. 2004; Wallace et al. 2006).

One reason for this is that dense and long time series of imagery at resolutions fine enough to study dryland vegetation change (e.g., Landsat-type images or finer) were until recently not available for larger areas (or only at high costs). The entire USGS Landsat archive has recently entered the public domain, providing exciting new possibilities to advance dryland mapping. A second reason is that there is no consensus regarding the choice of vegetation estimate to measure gradual change in drylands using time series of images, and both vegetation indices (Wallace et al. 2004; Washington-Allen et al. 2006; Paudel and Andersen 2010) and spectral mixture analysis (SMA) have been used to quantify vegetation cover (Hostert et al. 2003a; Röder et al. 2008b) although these measures may differ substantially.

SMA assumes that each image spectrum is a linear combination of a few pure spectra, so-called *endmembers* (Smith et al. 1990). Assuming linear mixture of endmembers, SMA fractions can directly be interpreted as percent ground coverage. SMA outperforms qualitative measures such as vegetation indices, particularly where vegetation is sparse (Elmore et al. 2000) and these advantages have been the main reason for selecting SMA to characterize vegetation for subsequent trend analyses. Yet, implementing an SMA-based approach over large areas is challenging because of dryland heterogeneity. The size of spectrally pure image objects is often much smaller than the spatial resolution of the sensor, which in turn often precludes the identification of an appropriate set of image endmembers, i.e. endmembers extracted from pixel spectra. Moreover, endmember variability can greatly increase with the size of the study region. Conversely, choosing a universal set of reference endmembers (i.e. spectra from a spectral database) that would be required for multi-date SMA (Elmore et al. 2000; Camacho-De Coca et al. 2004), is often not feasible. This is because (a) spectral libraries do not exist for many regions of the world and (b) transferring endmembers in time and space requires converting images to surface reflectance in most cases (Kuenzer et al. 2008), which can be challenging. Although SMA outperforms other vegetation estimates for single-date applications, the trade-off between using simpler estimates such as vegetation indices or Tasseled Cap Greenness for mapping land cover modifications from Landsat time series remains untested.

Our goal here was to assess the effect of choosing different vegetation estimates for Landsat-based trend analyses in drylands. We tested this for a study region on Crete (Greece), because this region offers a range of different dryland ecosystems, is characterized by highly complex, fine-scale landscapes where the advantages of SMA should be strongest (Hostert et al. 2003a). Specifically, our objectives were to

(1) derive four different vegetation cover estimates (SMA green vegetation fraction, NDVI, SAVI and Tasseled Cap Greenness) for a near-annual time series of Landsat images,

(2) normalize vegetation estimates and carry out trend analyses for all vegetation estimates, and

(3) compare resulting trends for all vegetation estimates and quantify differences among them.

2 Study region

The study region in Crete, Greece (Fig. II-1), encompasses the Messara Plain and the Psiloritis and Asterousia mountain ranges, resulting in a strong elevation gradient from sea level to almost 2,500 m above sea level (asl). Climate is sub-humid to semi-arid Mediterranean, with hot and dry summers and mild, relatively moist winters. Average temperature is 28 °C in summer and 12 °C in winter at sea level, but temperature varies substantially with elevation (Chartzoulakis et al. 2001). Limestone dominates the Psiloritis Mountains and the southern Asterousia Mountains, whereas Flysch is prevalent in the northern Asterousia Mountains. The Messara valley is an alluvial plain mainly composed of quaternary deposits.

Crete, like many other regions in the Mediterranean Basin, is dominated by cultural landscapes with long land use histories. Today's land use is mainly determined by topography and climate. Plains are intensively farmed with olive orchards and vineyards. Where relief precludes agriculture, natural and semi-natural vegetation communities dominate, most importantly phrygana (evergreen shrubs) and matorral (tall shrubs and degraded forests). These areas are used as rangelands, mainly for sheep and goat grazing, although some small-scale farming exists. After Greece joined the European Union in 1981, subsidies under the Common Agriculture Policy became available to Greek farmers. In the past, subsidies for livestock husbandry were scaling with the number of animal heads, resulting in starkly increasing sheep and goat numbers, often exceeded carrying capacity (Papanastasis 1998). As a result, rangeland vegetation in Central Crete has been substantially modified (Lyrintzis 1996; Hostert et al. 2003b). Thus, the study region encompasses a wide range of environmental conditions, has complex, heterogeneous landscapes, and is characterized by overgrazing, making it an ideal test site for testing the effect of choosing different vegetation indicators for trend analyses. We focused on rangelands solely, because we were interested in mapping subtle vegetation cover change only.

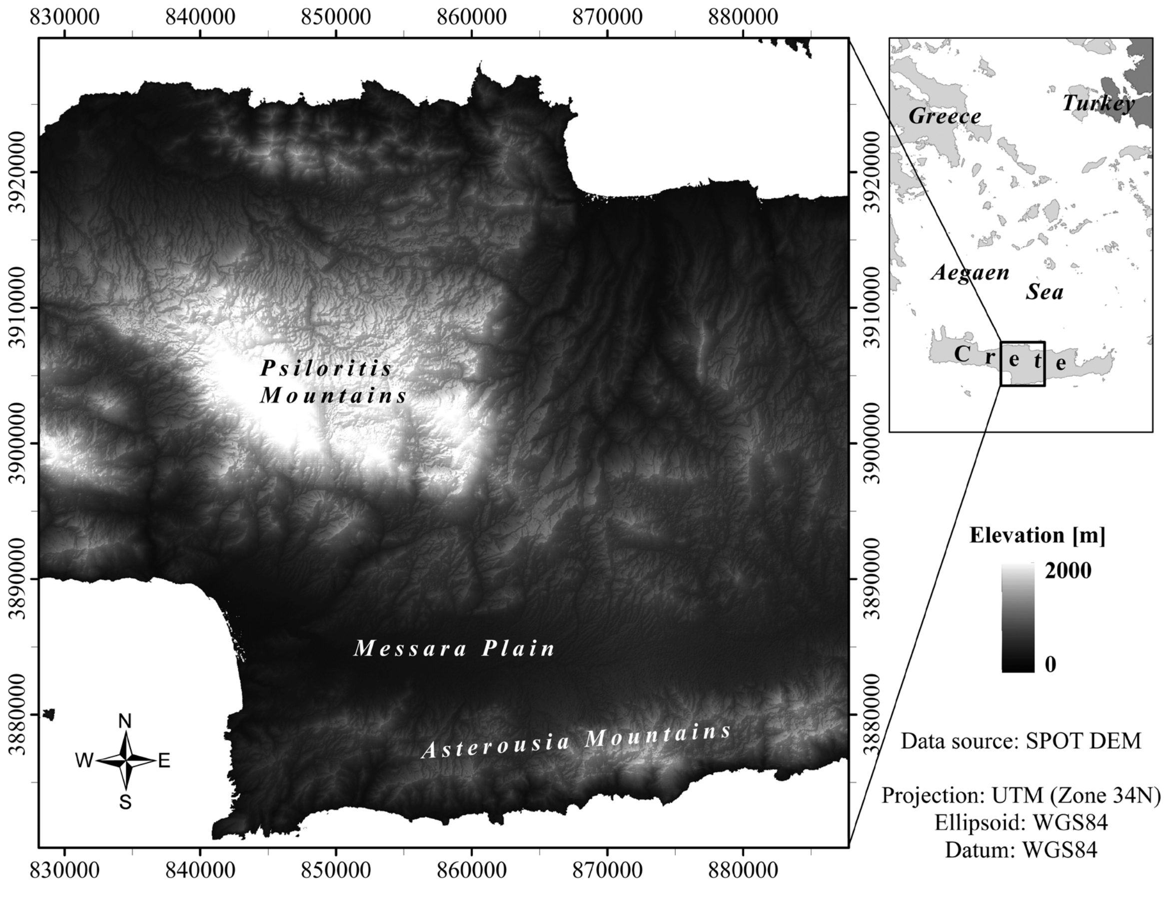

Figure II-1: The study area on Crete, Greece. Source: SPOT digital elevation model; ESRI World Data and Maps Kit 2005 (national boundaries).

3 Data

We acquired a near-annual time series of Landsat 5 TM and Landsat 7 ETM+ images from 1984 to 2005. Only images from mid-May to mid-June, the period of maximum vegetation vigor, and with low cloud cover were considered. Fourteen images from different years satisfied these prerequisites, resulting in a time series of at least one image every two years (Table II-1). Additionally, a broad-scale 10-day composite NDVI time series for the period 1989 to 2005 was available from the 'Mediterranean Extended Daily One Km AVHRR Data Set' (MEDOKADS) (Koslowsky 1998) to evaluate the phenological status of each Landsat image (Stellmes et al. 2010). Comparing the Landsat acquisition dates with these NDVI time series confirmed the good agreement of Landsat image acquisition dates and the vegetation peak for the years studied (Fig. II-2).

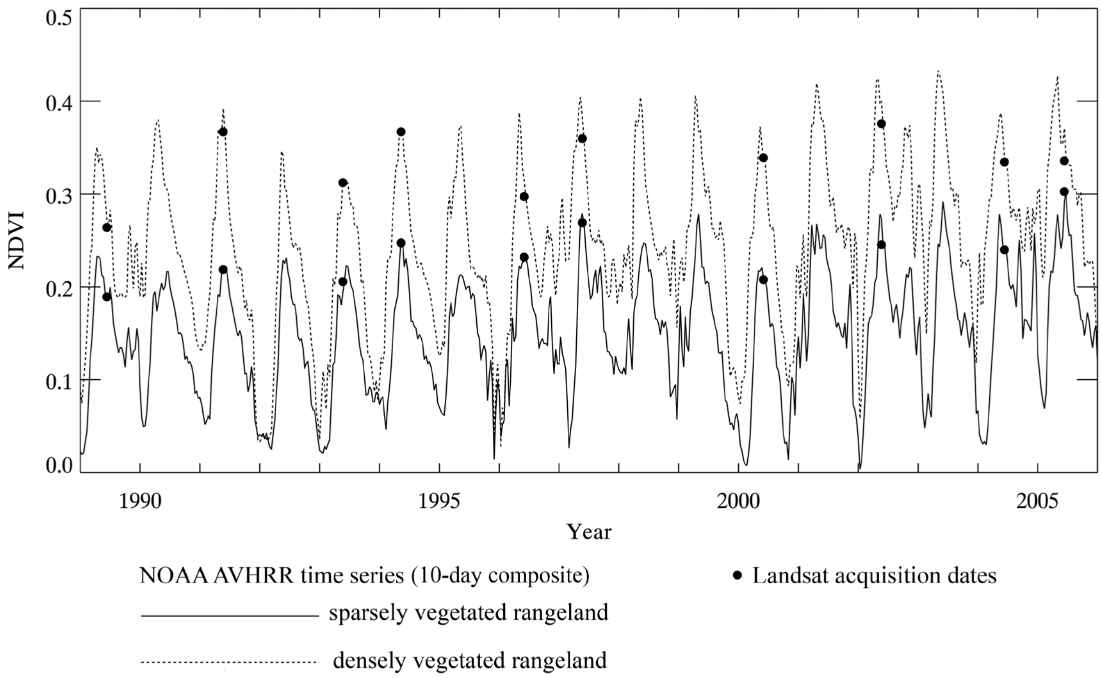

Figure II-2: NDVI time series from the MEDOKADS archive for two exemplary rangeland areas covering the time period 1989-2005. The acquisition dates of the Landsat time series are marked with circles for the overlapping time span.

Ten of the Landsat images were available from a previous study and had already been ortho-rectified (UTM, Zone 34N, WGS 84) (Hostert et al. 2003a). The four additional TM and ETM+ images were co-registered to the existing time series using the 1994-image as reference and semi-automatic tie point search (Roberts et al. 1998). To correct for relief displacement, we incorporated a digital elevation model (DEM) derived from a stereoscopic pair of SPOT images (spatial resolution of 20 m). All images were ortho-rectified with high positional accuracy and overall root-mean-squared errors < 0.35 pixels (~11 m).

Table II-1: Acquisition dates of Landsat images used

Acquisition date	*Sensor*
06/03/1984	TM
05/24/1986	TM
06/12/1987	TM
05/29/1988	TM
06/17/1989	TM
05/22/1991	TM
05/27/1993	TM
05/30/1994	TM
06/04/1996	TM
05/22/1997	TM
05/30/2000	TM
05/28/2002	ETM+
06/10/2004	TM
06/13/2005	TM

We assembled an extensive spectral library of vegetation, soil, and rock spectra for the study area. Fresh leaf-stacks, soil, and rock samples were collected in the field during spring and summer of 1996, 1998 and 2005. Sampling was performed considering geological, soil, and land use maps to cover the variety of strata in our study region. All samples were measured under laboratory conditions using an ASD FieldSpec Pro II spectroradiometer covering the wavelength range of 350-2,500 nm at a spectral resolution of 3.0 nm. We also collected *in-situ* measurements of all major vegetation species thereby integrating the spectral signal of the entire canopy (leafs, stems, and branches). All spectra were resampled to match the spectral resolution of the Landsat TM and ETM+ bands. In total, our spectral library contained 50 vegetation spectra (30 leaf spectra measured in the lab and 20 *in-situ* measurements), 30 soil spectra, and 30 rock spectra.

To allow comparing reference spectra and Landsat imagery, all images were converted to surface reflectance following the pre-processing scheme developed in previous work (Hostert et al. 2003a). First, raw images were converted to at-satellite radiances using calibration gain and offset parameters (http:/landsat7.usgs.gov/science_L7_cpf.php ; Canty and Nielsen 2008). We used a modified radiative transfer model based on the 5 s Code (Garcia-Haro et al. 1996) to convert radiance to surface reflectance, integrating the DEM to account for illumination effects (Hill et al. 1995). We chose the 1994- image as a reference image due to its high radiometric quality and its central position in the time series and adjusted input parameters iteratively based on the reflectance of known invariant features (i.e., limestone, dense urban settlements, and water) (Schott et al. 1988; Hostert et al. 2003a). Variations of target spectra in time did not exceed an RMSE of 0.02 for all images suggesting a high radiometric consistency of the time series.

We obtained the CORINE land cover 2000 map for Crete (http://dataservice.eea.europa.eu). This 1:100,000 map includes 44 land cover classes and has a minimum mapping unit of 25 ha. Because our focus was on rangelands, we used the CORINE map to mask all 'semi-natural areas'. We also digitized all clouds, cloud shadows, and snow-covered areas in all images.

4 Methods

4.1 Vegetation estimates

To assess how the choice of a particular vegetation estimate affects trend analyses, we calculated three different types of vegetation estimates: (1) vegetation indices (NDVI and SAVI), (2) Tasseled Cap Greenness, and (3) SMA-based fractional vegetation cover.

4.1.1 Vegetation indices

We calculated the NDVI because it is the most widely used vegetation index. The NDVI is a ratio-based index capturing the spectral contrast between the red and near-infrared (NIR) reflectance (ρ) of the vegetation signal and is defined as:

$$NDVI = \frac{\rho_{NIR} - \rho_{RED}}{\rho_{NIR} + \rho_{RED}} \tag{1}$$

However, NDVI is affected by soil spectral properties, especially where vegetation cover is low. This may result in a relative over-estimation of vegetation on dark soils, a relative under-estimation of vegetation on bright soils, and NDVI usually results in non-zero although vegetation may not be present (Huete et al. 1985). To compensate for these shortcomings, several alternative vegetation indices have been proposed. Here, we used the Soil Adjusted Vegetation Index (SAVI) that is based on the soil-line concept (the soil line is a linear relationship between the red and NIR bare soil reflectance values) to reduce background influence and that has been specifically developed for environments with sparse vegetation (Huete 1988). Technically, the SAVI is a modification of the NDVI incorporating an additive term L to correct for soil brightness influence:

$$SAVI = \frac{\rho_{NIR} - \rho_{RED}}{(\rho_{NIR} + \rho_{RED} + L)}(1 + L) \tag{2}$$

L depends on the proportional vegetation cover as well as the vegetation density. For very sparse vegetation or bare soils, L approximates one, whereas L converges to zero in densely vegetated areas (in this case the SAVI and NDVI become equal). We set L = 0.5, as suggested by Huete (1988) for reducing the soil influence in sparsely vegetated areas.

4.1.2 Tasseled Cap Greenness

The Tasseled Cap is a linear transformation of the multi-dimensional Landsat spectral space, yielding orthogonal axes that contain the majority of the spectral information. Originally developed to better understand crop phenology in four-dimensional Landsat Multi Spectral Scanner (MSS) images, the Tasseled Cap transformation has been adapted to Landsat TM data (Hill and Schutt 2000). The first three components of the Tasseled Cap transformation for Landsat TM can be interpreted thematically, representing brightness, greenness and wetness. We used the coefficients for reflectance data (Crist 1985) and extracted the greenness component for each image in the time series.

4.1.3 Spectral mixture analysis

Spectral mixture analysis is based on the assumption that each pixel spectrum is a linear combination of the spectra of a few, spectrally distinct components, so-called endmembers (e.g., vegetation, soil, rock, etc.) (Johnson et al. 1983; Roberts et al. 1993). The fraction of each endmember per pixel can then be derived by solving the equation:

$$\rho_j = \sum_{i=1}^{n} f_i * \rho_{i,j} + e_j \tag{3}$$

Here, ρ_j denotes the reflectance in band j, *n* stands for the number of endmembers, f_i is the fraction of endmember *i*, $\rho_{i,j}$ denotes the reflectance of an endmember spectrum of endmember *i* in band *j*, and e_j is the band-wise residual. Model fit is assessed by summing the band-wise residuals over all spectral bands and calculating the root-mean-squared error (RMSE) for each pixel:

$$RMSE = \sqrt{\sum_{j=1}^{n} \frac{e_j^{\,2}}{n}} \tag{4}$$

In our case, the fractions of all endmembers are subject to the unity constraint, assuring that each pixel's abundances sum to one. Mathematically, the number of endmembers is restricted to the number of bands plus one. However, collinearity in Landsat TM/ETM+ spectral feature space limits the number of endmembers to the inherent spectral dimensionality and a maximum of three to four endmembers is usually supported by Landsat data (Adams et al. 1995; Small 2004).

We selected the 1994-image as our reference image for fitting the initial spectral mixture model for our study area. We MNF- (Minimum Noise Fraction) transformed the Landsat feature space to reduce the spectral dimensionality of the image in order to simplify the selection of endmembers. While the eigenvalues suggested a five-dimensional spectral feature space, the 5th MNF component represented only a small area with different bedrock than the rest of the study region (Fig. II-3). We therefore assumed a four-dimensional feature space, similar to what was previously suggested for a subset of our study region (Hostert et al. 2003b). To select candidate endmembers from our spectral library, we applied the forward MNF transformation to all reference spectra and subsequently plotted them into the transformed Landsat feature space. Ideally, endmembers are located at the boundary of the pixel cloud to maximize spectral contrast between endmembers and to avoid fractions exceeding one. Using different combinations of two-dimensional feature space plots, we identified three photosynthetic active vegetation spectra (hereafter *green vegetation*), four soil spectra, and three bedrock spectra that satisfied these criteria and constituted candidate spectra (Fig. II-3).

We then fitted SMA models to all possible combinations of candidate endmembers, resulting in 36 different four-endmember models. To account for brightness differences and remaining shading effects, we included a shade endmember (i.e., zero reflectance) in all models. We then compared all models based on the spatial pattern and overall mean RMSE, the residual images and histograms of endmember fractions (Adams et al. 1995; Röder et al. 2008b).

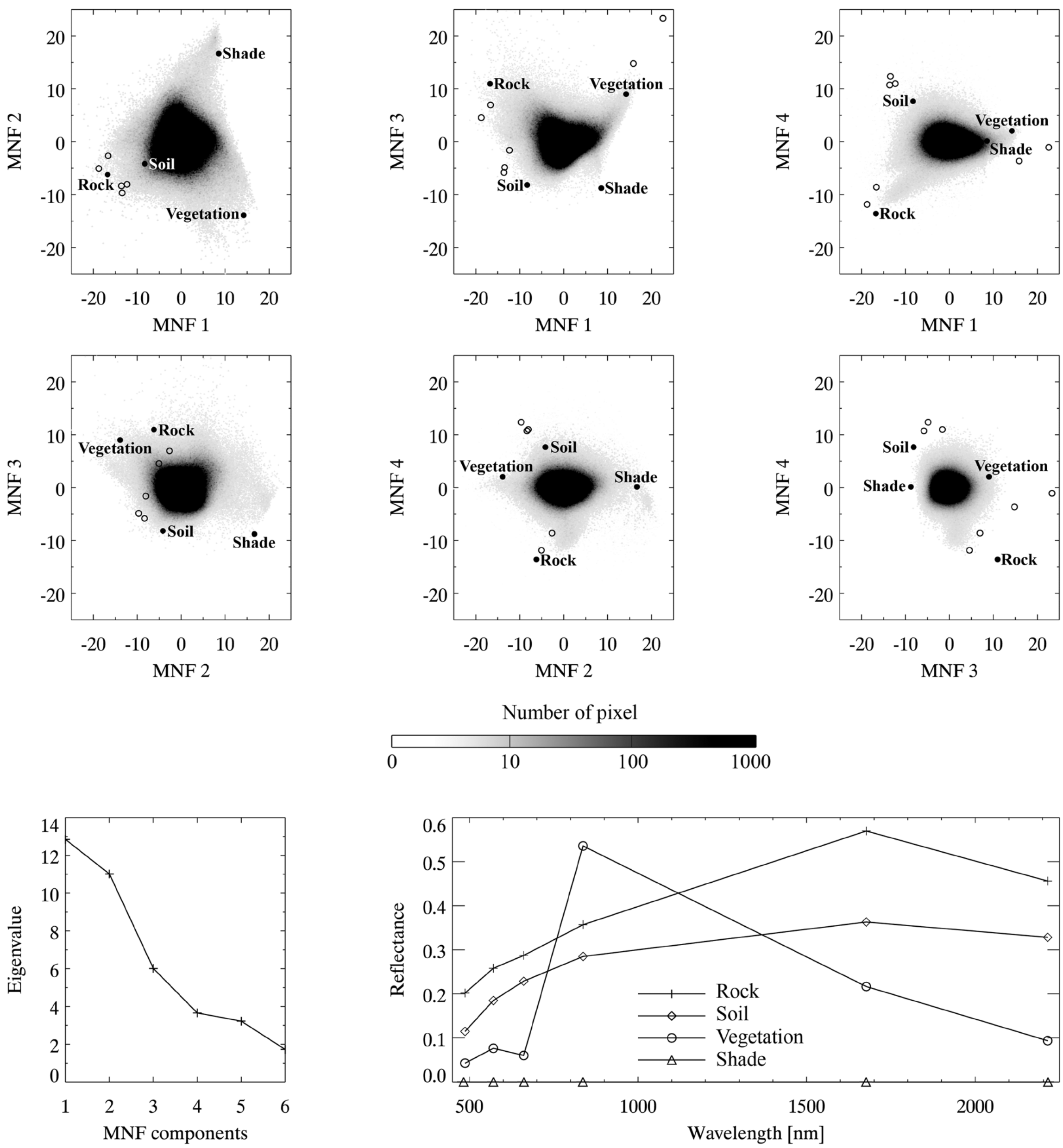

Figure II-3: MNF components of the Landsat feature space and endmember spectra. The distribution of eigenvalues of the MNF component rotation shows a general high spectral dimensionality of the study region. A subset of candidate endmember spectra is overlaid to the spectral feature space of the MNF components. Here, filled circles represent the endmembers of the SMA model whereas the non-filled circles show considered but not selected candidate endmembers. The endmember spectra of the final model consisted of green vegetation, soil, limestone and shade.

Our best SMA model contained a Thorny Burnet (*Sarcopoterium spinosum*) integrated shrub spectrum, a chromic cambisol soil spectrum, a limestone spectrum, and shade (Figure 3). For this model, band-wise residuals did not show systematic spatial patterns and the RMSE in general did not exceed 2% reflectance for all pixels. In addition, the spatial distribution of endmember fractions for this model was consistent with expert knowledge on study site characteristics. Endmember fractions generally ranged between zero and one, however, slightly negative green vegetation fractions occurred for the different bedrock area not represented by our SMA model, slightly negative soil fractions were found for pure limestone likewise. We did not redistribute the shade component among the other endmember fractions because it was correlated with the limestone fractions which would have led to a bias in vegetation cover in these areas.

Transferring SMA models among multi-date imagery requires a universal set of endmembers (Elmore et al. 2000; Camacho-De Coca et al. 2004). Once a model was fitted to the reference image, we applied this model to all images in the time series. Using an identical endmember set ensures stable endmember fractions in unchanged areas, and changes in fractions among images therefore likely represent true changes in surface components (Elmore et al. 2000; Camacho-De Coca et al. 2004). In our study, the temporal variation of endmember fractions of pseudoinvariant features was less than 0.04 for the green vegetation fraction, suggesting a high consistency among images in the time series (Fig. II-4).

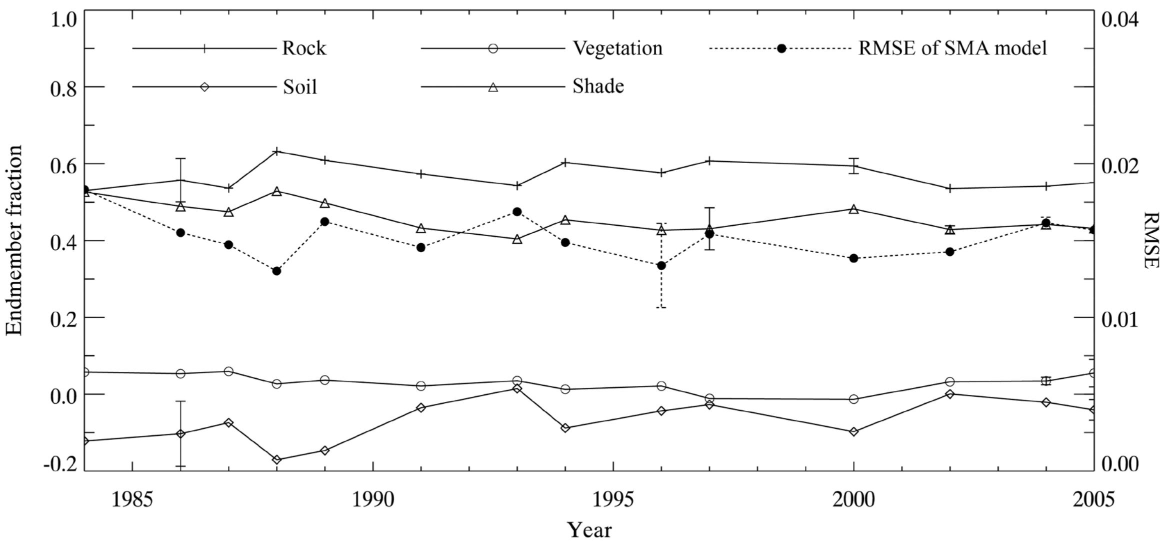

Figure II-4: Temporal variation of endmember fractions for pseudo-invariant features (limestone) within the study region (mean value with error bars indicating years with lowest and highest standard deviation)

4.2 Normalization of vegetation cover estimates

SMA quantifies fractional green vegetation cover whereas NDVI, SAVI and Tasseled Cap Greenness are dimensionless. The different ranges and scales of the vegetation estimates make comparisons among them difficult and hinder contrasting the results of subsequent trend analyses. To allow for direct comparisons of trends based on different estimates, we normalized all estimates using a z-transformation by subtracting the mean of each population of each estimate and dividing the difference by the standard deviation (Table II-

2). Thus, z-scores represent the difference between the original value and the vegetation estimators population mean in units of the standard deviation. It is important to note that this approach does fully retain the overall distribution of vegetation cover estimates and the temporal profiles of vegetation estimates for a given pixel (i.e., the trend analyses on normalized and un-normalized data are identical). In the following, we refer to z-normalized vegetation estimates as SMA_z, $NDVI_z$, $SAVI_z$ and TC_{Gz}, respectively.

Table II-2: Mean and standard deviation of the time series for each vegetation estimate. These parameters were utilized for the z-transformation.

Vegetation estimate	*mean*	*standard deviation*
SMA	0.33	0.16
NDVI	0.33	0.12
SAVI	0.22	0.08
TC_G	0.07	0.04

4.3 Trend analysis

To calculate vegetation cover trends for different vegetation estimates, we applied a pixel-wise linear trend function. Linear trend functions have been shown to effectively capture trends in rangeland vegetation, both in our study area (Hostert et al. 2003a) and other Mediterranean rangelands (Röder et al. 2008a).Linear trends were fitted to the time series of SMA abundances and to the z-normalized vegetation estimates (SMA_z, $NDVI_z$, $SAVI_z$ and TC_{Gz}) for each pixel to determine the spatial patterns of trend directions and magnitudes and to quantify differences between the trends of the four vegetation estimates:

$$y(t) = a * t + b \tag{5}$$

Regression constants (b) and coefficients (a) were calculated using a least-square fit and the Landsat acquisition dates (t) as independent variable. Areas with clouds were disregarded in the calculation of trends. We expressed t in days after the acquisition date of the oldest Landsat image (3rd June 1984, $t_0 = 0$), which allowed correctly considering the time lag between subsequent image dates. Thus, b represents the initial vegetation cover in 1984, while a determines the direction and magnitude of vegetation cover change per pixel and day. We rescaled the dimensionless trend coefficients for all methods by multiplying them with the maximum number of days in our time series ($t_{13} = 7{,}679$). This allowed us to derive absolute vegetation cover change across the entire observation period for each pixel and for all vegetation estimates. The strength and goodness-of-fit of the pixel-wise linear trend models was assessed by calculating the correlation coefficient (r) and the RMSE. We tested the significance of the trend coefficients using a two-sided t-test.

To compare trend coefficients among the different vegetation estimates, we generated scatterplots between the SMA_Z-based trend coefficients and the trend coefficients of the qualitative methods ($NDVI_Z$, $SAVI_Z$ and TC_{Gz}). Additionally, we calculated Pearson's R^2 for each estimate pair ($SMA_Z/NDVI_Z$, $SMA_Z/SAVI_Z$ and SMA_Z/TC_{Gz}) and derived mean absolute and standard deviation for every population of coefficients. We plotted differences in trend results of each estimate pair against the initial vegetation cover to further analyze the underlying mechanism of differences in trend estimates. Likewise, we plotted differences in trend results against the trend magnitude and we also calculated difference images between vegetation trends based on different estimates.

5 Results

Trend analysis of SMA-derived vegetation estimates showed that most of the rangelands in the study region were characterized by relatively low vegetation cover by the beginning of the observation period (Fig. II-5). About two thirds of all pixels showed SMA vegetation fractional cover of 10-40% in 1984. Vegetation cover changed substantially in Crete during the 21-year (1984-2005) time period we studied, with an overall decrease of 4.5% at the study region level. These dynamics were mainly due to gradual vegetation changes (i.e., the trend magnitude for almost all pixels (> 96%) ranged between +/- 25%), whereas sudden vegetation cover changes were scarce and showed partially high trend magnitudes (> 25% vegetation cover change). Gradual changes were accompanied by relatively low to high correlation coefficients (mean value of |r| was 0.37 with a standard error of 0.21) and low RMSE values. In contrary, abrupt changes in the temporal profiles showed to some extent high |r| values as well but also higher values in the RMSE image. The spatial pattern of vegetation change was highly heterogeneous with co-occurring areas of vegetation increase and vegetation decrease (Fig. II-5). Vegetation trends also differed markedly between the two mountain ranges. Large patches of declining vegetation characterized the Asterousia Mountains in the South of the study region and only a few areas showed vegetation increase. This was different in the Psiloritis Mountains that were mainly characterized by large patches of stable or increasing vegetation cover (e.g., the sub-alpine areas in the center of the study region, Fig. II-5). Smaller rangeland patches scattered within the cropland matrix in the plains did not exhibit particular spatial patterns in the vegetation cover trends.

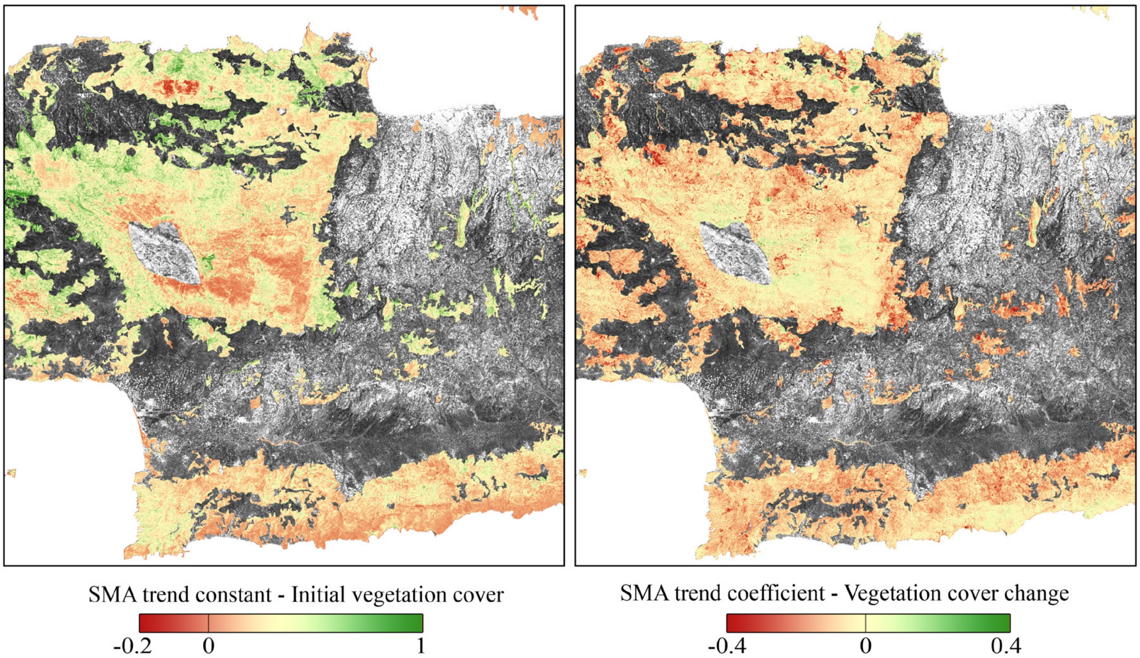

Figure II-5: SMA derived vegetation cover change in Central Crete for 1984-2005 based on the trend coefficient (right) and the initial vegetation cover depicted by the trend constant (left).

The relationship between SMA fractions and the other vegetation estimates was linear, with R^2-values of 0.77 (SMA/NDVI), 0.89 (SMA/SAVI) and 0.94 (SMA/TC_G) and equal values for the z-normalized estimates (Fig. II-6). Scattergrams of z-normalized values of the vegetation estimates vs. the SMA vegetation fraction closely matched the 1:1-line for all measures with mean absolute biases well below 0.4 (0.37 for $NDVI_z$, 0.26 for $SAVI_z$ and 0.19 for TC_{Gz}). Plotting mean absolute differences between each estimate pair and the corresponding standard deviation against SMA_z-derived vegetation cover revealed marked differences among the three normalized vegetation estimates. Mean differences between $NDVI_z$ and SMA_z fractions showed high values at very low and high vegetation cover level with standard deviation increasing likewise. $SAVI_z$ showed a similar deviation pattern but overall less influenced than $NDVI_z$. TC_{Gz} was the least affected among the vegetation estimates. Differences were almost constant for high vegetation cover levels but increased with decreasing vegetation cover.

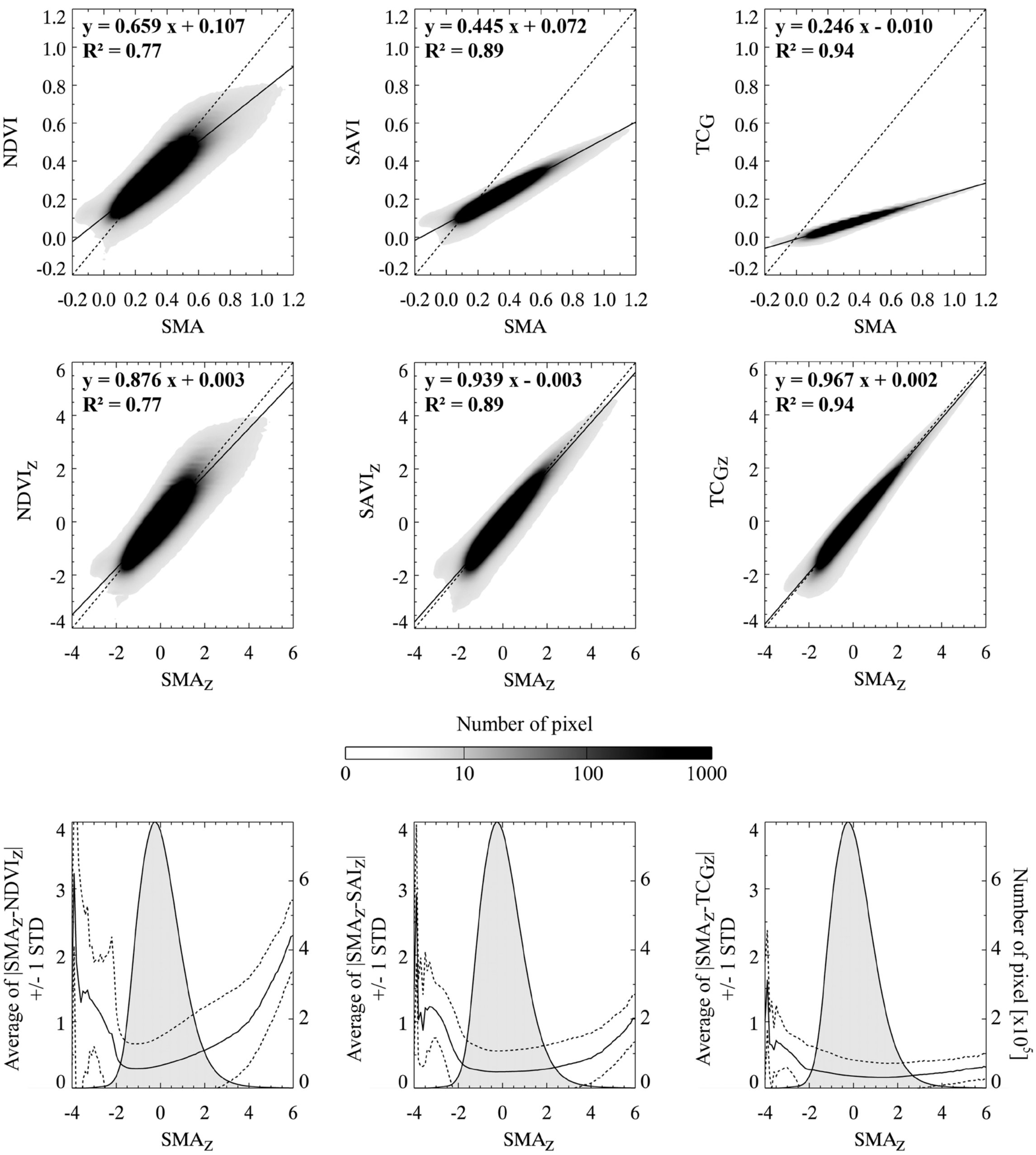

Figure II-6: Top: Scatterplots between SMA-based fractional vegetation cover and the three vegetation estimates before normalization (NDVI/SMA, SAVI/SMA, and TC_G/SMA). Middle: Scatterplots between SMA-based fractional vegetation cover and the three vegetation estimates after normalization ($NDVI_z/SMA_z$, $SAVI_z/SMA_z$, and TC_{Gz}/SMA_z). Bottom: Differences between SMA_z fractional vegetation cover and normalized vegetation estimates across different levels of vegetation cover (solid line = mean absolute residuals, dashed lines = standard deviation of residuals, the histogram of fractional vegetation cover in the study region is overlaid in grey).

Comparing among vegetation trends based on different vegetation estimates revealed that all methods resulted in comparable spatial patterns of vegetation dynamics for our study region. This was similar for both low and high trend magnitudes, and for different initial vegetation cover levels. We also did not find any systematic bias when comparing among our maps (Fig. II-7).

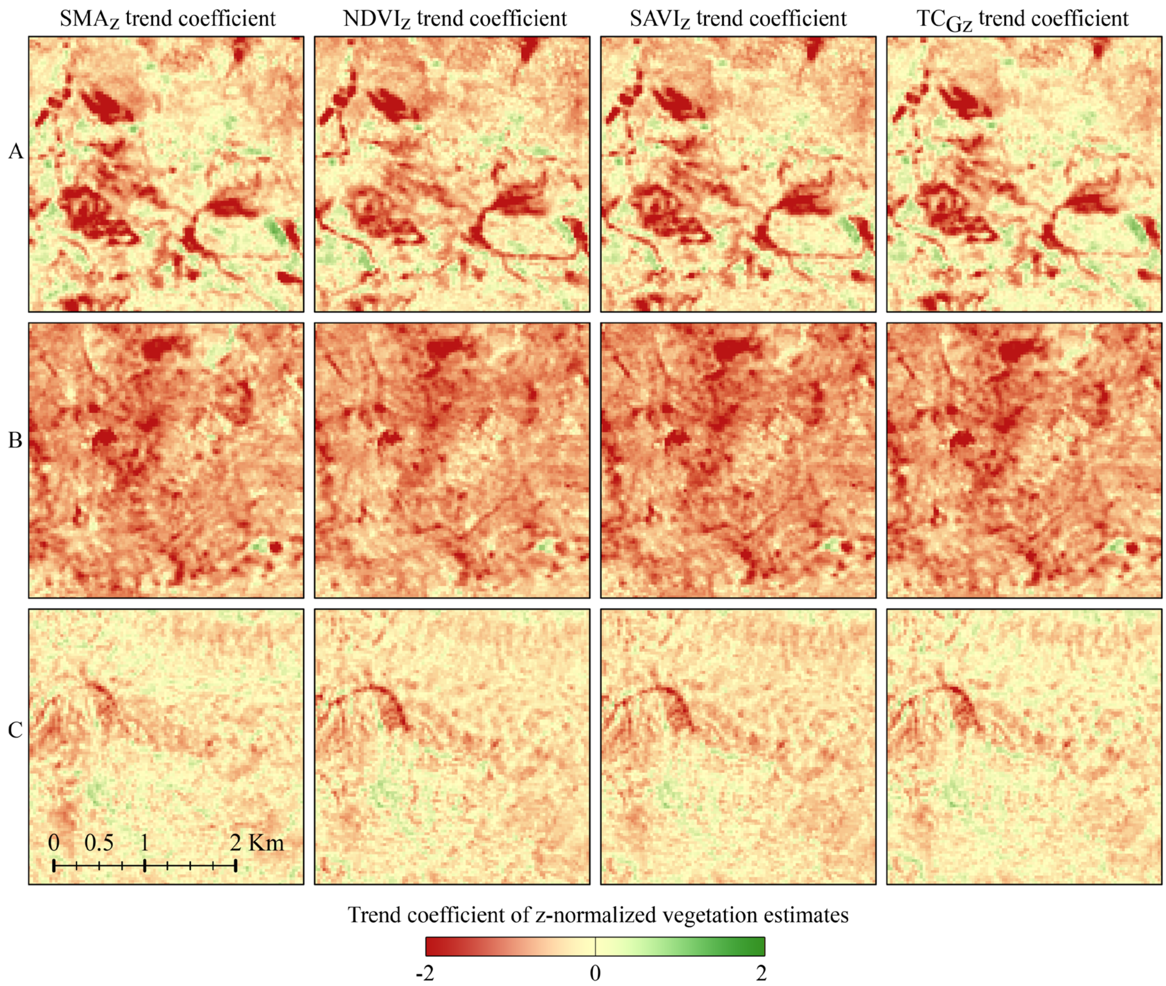

Figure II-7: Examples of trend coefficients based on the z-normalized vegetation estimates (SMA_z, $NDVI_z$, $SAVI_z$, TC_{Gz}) across the study area and three different examples (A ,B ,C).

Quantifying differences among vegetation trends based on different estimates confirmed the high level of agreement between the four methods (SMA_z, $NDVI_z$, $SAVI_z$ and TC_{Gz}). Particularly, the qualitative estimates were highly similar to the quantitative measure of SMA fractional cover (Fig. II-8, top). Scatterplots of trend coefficients (SMA_z/$NDVI_z$, SMA_z/$SAVI_z$ and SMA_z/TC_{Gz}) were all close to the 1:1-line and mean absolute biases of trend coefficients ranged well below 0.3 z-value of vegetation cover (0.29 for $NDVI_z$, 0.20 for $SAVI_z$ and 0.14 for TC_{Gz}, respectively).

Mean and standard deviations of absolute trend differences between two methods depended on trend magnitude (Fig. II-8, middle). Mean absolute differences were generally lower for small trend magnitudes and increased as trend magnitude increased for the three method pairs (Fig. II-8, middle). This increase was higher for $NDVI_z$-based trends compared to the other methods. Similarly, the variability of differences was also higher for $NDVI_z$-based trends. Mean differences for TC_{Gz}-based trends were generally lower for low trend magnitudes than for $NDVI_z$- and $SAVI_z$-based trends. TC_{Gz}-based trends exhibited the lowest variability of differences with varying trend magnitudes (Fig. II-8, middle).

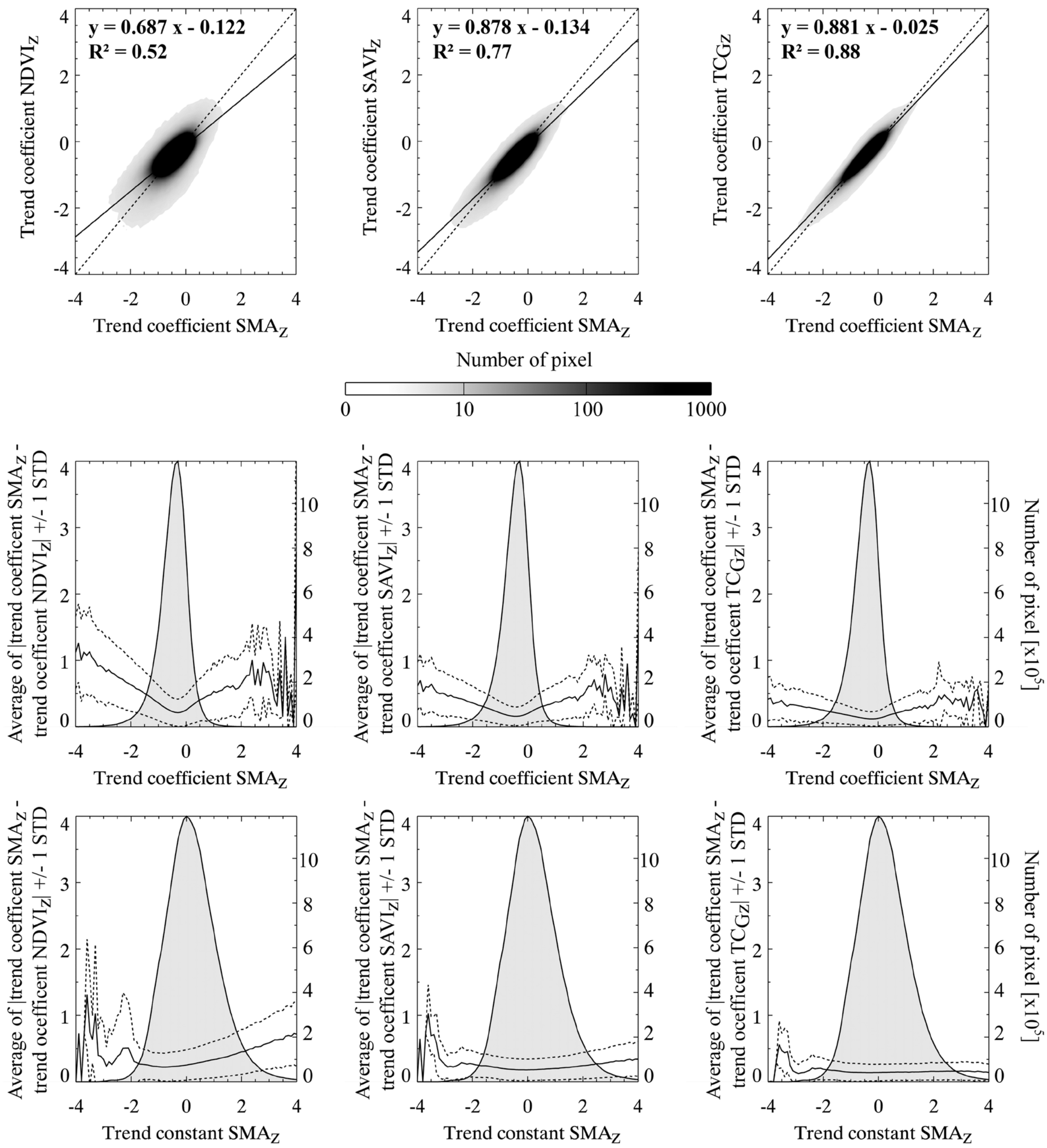

Figure II-8: Top: Scatterplots of z-normalized SMAz-based trends vs. trends based on qualitative vegetation estimates. Middle: Mean absolute differences between z-normalized SMAz-based trends and NDVIz-, SAVIz- and TCGz-based trends across different levels of trend magnitude (solid line = mean absolute differences, dashed lines = standard deviation of absolute differences, the histogram of trend magnitudes is overlaid in grey). Bottom: Differences between z-normalized SMAz-based trends and NDVIz-, SAVIz- and TCGz-based trends across different levels of initial vegetation cover (solid line = mean absolute differences, dashed lines = standard deviation of absolute differences, the histogram of trend magnitudes is overlaid in grey).

Differences between trend results also depended differently on the vegetation cover (Fig. II-8, below). Whereas TC_{Gz}-based trends were unaffected by varying levels of the initial vegetation cover level, mean absolute differences between the $NDVI_z$- and $SAVI_z$-based trend results were low for the mean vegetation level (coefficient constants with corresponding z-value of 0) but increased for higher and lower vegetation values. This increase was more pronounced for $NDVI_z$- than for $SAVI_z$-based trends.

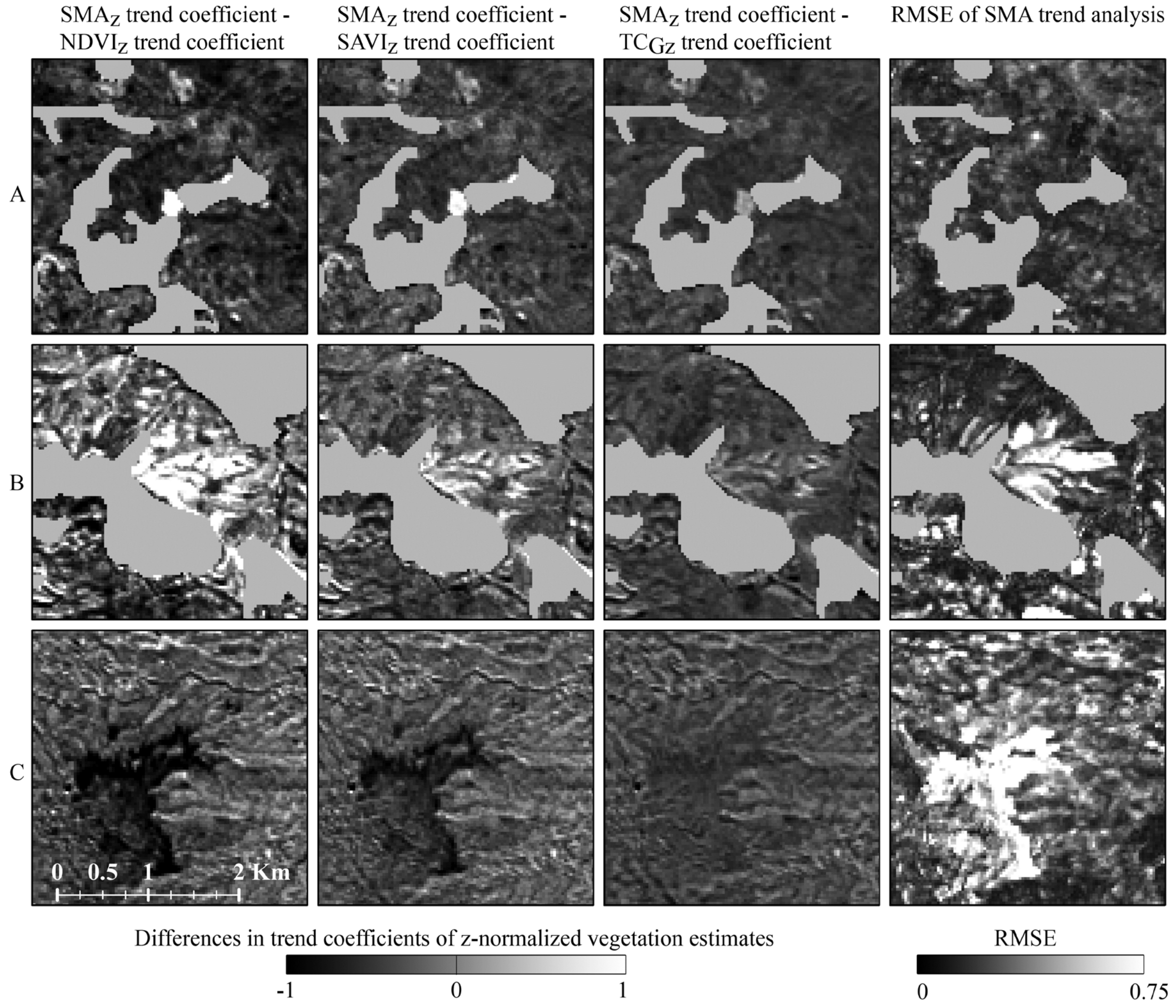

Figure II-9: Examples of high differences in trend results based on the z-normalized vegetation estimates (SMA_z, $NDVI_z$, $SAVI_z$, TC_{Gz}) across the study area for three different areas (A, B, C). The RMSE of the linear trend analysis is additionally included.

Differences between trends calculated using different vegetation estimators generally did not show distinct spatial patterns, while some patches of high and low differences were found (Fig. II-9). These patches of consistently negative or positive trend differences were mainly found where abrupt disturbances had occurred during the observation period. Most of the disturbances were related to land cover conversions, especially clearings of the semi-natural vegetation. This process was particularly evident in the Asterousia Mountains where farmed areas expanded in formerly sparsely vegetated semi-natural areas close to farmland fringes (Fig. II-9, A). High differences in trend results were related to high RMSE values of the SMA after clearings, indicating that the SMA model did not well represent these post-conversion land cover types (Fig. II-10, A). High differences were also partially accompanied by high RMSE values of the linear trend analysis (Fig. II-9, B) indicating non-linearity. This was evident for fire events in former dense vegetation canopies and the subsequent recovery process. While vegetation was registered similarly by all vegetation estimators before the fire, estimates varied during the subsequent recovery process. RMSE of the SMA showed high values directly after a fire event and only a slow decrease in the following years suggesting the mixture model did not represent burnt land covers (Fig. II-10, B). Finally, high differences also appeared where canopies showed high phenological dynamics between years. This was accompanied by high RMSE values of the linear trend analysis indicating the high variability of the individual estimates throughout the time series (Fig. II-10, C).

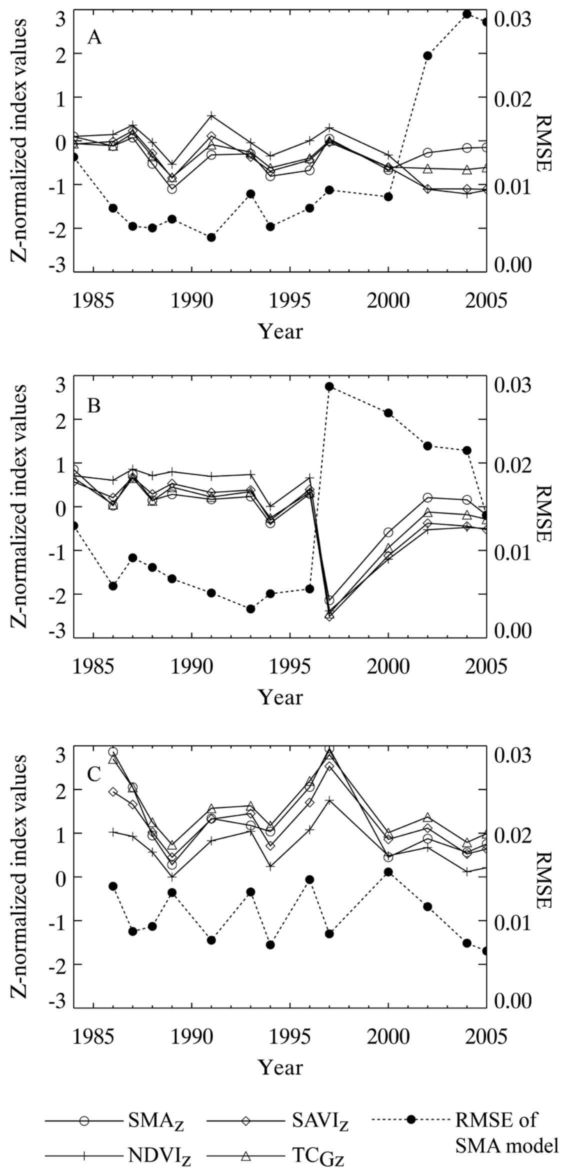

Figure II-10: Temporal profiles of the normalized vegetation indices and SMAz for the three examples (A, B, C) shown in Fig. II-9. Additionally, the RMSE of the SMA is included.

6 Discussion

Time series analysis is a powerful tool for mapping gradual land cover change, but it is unclear how the choice of vegetation estimate affects trend analyses. Our comparison of trends based on four different vegetation measures (Spectral mixture analysis, NDVI, SAVI and Tasseled Cap Greenness) showed that all methods resulted in highly comparable trends. Absolute trend differences between measures were less than 5% vegetation cover, similar to the uncertainty introduced by sensor noise (Hostert et al. 2003a). Trend analysis appears to reduce phenology influence, regardless of vegetation estimator used, compared to traditional change detection approaches based on a few image dates, but nevertheless requires a careful selection of phenologically comparable imagery. Our study clearly indicates that trend analyses may be relatively insensitive to the choice of a particular vegetation estimate. For most applications, this can be considered an advantage and being able to use simple, easily to derive indices with confidence opens the pathway to reconstructing fine-scale vegetation dynamics and land use histories in drylands across large areas. However, we caution that for some applications, especially for land degradation and soil erosion studies in low cover environments or when mapping disturbance events such as fires is the focus, the differences in trends we found may still be important.

SMA often outperforms qualitative vegetation measures in single-date analyses, especially where vegetation cover is sparse (Garcia-Haro et al. 1996; Elmore et al. 2000), and this has been the main reason for carrying out SMA-based time series analyses in the past. While we also found spatial differences in the vegetation estimates when focusing on individual dates, our results suggest that the advantages of SMA are diminishing in multitemporal trend analyses and mean absolute differences between trends were marginal. Three main reasons explain the high agreement among trends. First, the temporal information inherent in Landsat time series appears to largely compensate for existing differences in vegetation measures (i.e., differences among vegetation estimates are either captured in the residuals of the trend analysis or errors are consistent over time). This is particularly evident in the much higher variability of single-date differences among estimates (Fig. II-6) compared to trend differences based on the entire time series (Fig. II-8). Second, the relationship between SMA and qualitative vegetation estimates was linear in our case, similar to what has been found for other ecosystems characterized by sparse vegetation cover (Wittich and Hansing 1995; Xiao and Moody 2005). These linear relationships were strong for all estimates ($R^2 > 0.77$), suggesting that vegetation cover in our study region was well below the level where qualitative estimates saturate in responses to increasing leaf area index (Carlson and Ripley 1997). Third, our time series consisted of near-annual images, thereby reducing the influence of phenology and other environmental factors that different vegetation estimates may respond to differently.

Among the alternatives to the SMA, NDVI performed weaker than Tasseled Cap Greenness (NDVI trends differ from SMA trends by 5%, TCG by 2.5 %), adding support to recent work by Kennedy et al. (2010) who tested the suitability of different vegetation estimates to capture subtle trends in forest condition. While our study suggests that TC_G-based trend analyses allows for effective monitoring of gradual vegetation change in drylands, the inherent advantages of SMA are undisputed. These advantages include, for example, the quantitative nature of the approach resulting in vegetation abundances rather than index values, lower sensitivity to saturation effects at high vegetation cover, sensitivity to low vegetation cover, possibilities to include background variability (Roberts et al. 1998), and the direct accuracy assessment of model performance (RMSE). Especially if quantitative biophysical estimations are the goal, the usefulness of qualitative vegetation estimates is limited. However, we suggest that in cases where mapping relative vegetation change across larger areas is the goal (i.e., most land use change studies), the trade-off in using simple qualitative indices may be marginal.

Although trends from different vegetation estimates were overall highly similar in our study, some potential sources of inaccuracies of qualitative methods compared to SMA also became evident:

(1) Non-linearities between SMA estimates of vegetation and qualitative measures: Non-linear relationships between SMA and qualitative measures may result in systematic bias in trend coefficients, especially if vegetation cover is dense and indices saturate (Elmore et al. 2000). Particularly NDVI showed substantial variability at high vegetation cover (> 60%, Fig. II-6) and a stronger dependency on trend magnitude. Overall, this suggests that NDVI may be less suited for mapping land cover modifications in drylands than the other vegetation estimators in this study.

(2) Influence of the unexplained background signal: Qualitative greenness measures are often substantially affected by soil background reflectance (Huete et al. 1985). This likely explains the somewhat higher differences between normalized vegetation estimates and SMA fractions at low vegetation cover levels (Fig. II-6) which likely contribute to the higher differences in trends at high trend magnitudes.

(3) Dependency on land cover types: Land cover type exerted some influence on differences among SMA-based trends and qualitative measures, especially if a land cover class shows high phenological variability. For example, we found high trend differences for grasslands that are only a minor component of the rangelands in our study region. The high phenological variability likely affects a single year's relationship between different vegetation estimates and leads to varying trends. Moreover, trend confidence significantly decreases with high year-to-year variability. Related differences in trends were relatively small based on TC_{Gz} but high for $SAVI_z$ and $NDVI_z$-based trends. A solution would be to stratify the study region prior to trend analyses, requiring a-priori knowledge that is not always available.

(4) Response to sudden disturbances: Vegetation estimates also responded differently to sudden disturbances such as fires or land cover conversions (e.g., agricultural expansion). While overall scarce, such disturbances are mirrored in the RMSE of the SMA, and because SMA responds differently to non-vegetated areas and the gradual recovery process than other measures, result in relatively high differences among methods. Indeed, most of the larger patches depicting high deviations between methods were attributable to such phenomena and highlight the weakness of a static endmember model to account for a change in land cover composition (Fig. II-10). Our focus here was exclusively on gradual vegetation change (e.g., due to shifting grazing regimes) that represented the vast majority of vegetation changes in our study region, but it is important to note that time series analyses can be extended to map and separate sudden disturbances from gradual changes, for example using trajectory-based approaches (Kennedy et al. 2007; Kennedy et al. 2010) or by adapting local models (Röder et al. 2008a).

In our analyses we tested the trade-off involved when using qualitative vegetation measures in time series analyses rather than SMA vegetation fraction, thereby assuming that the latter closely matches true vegetation cover. RMSE, model residuals, and visual assessment of SMA error distribution all do not suggest a systematic bias. Yet, a few factors may have contributed uncertainty to the SMA, including nonlinear mixing of sparse vegetation and soil (Huete et al. 1985) and the fixed set of endmembers we used. Stratifying the study area prior to the SMA (Camacho-De Coca et al. 2004; Garcia-Haro et al. 2005) or using endmember bundles (Roberts et al. 1998) are potential solutions to these problems. Moreover, the normalization of the shade fraction might change the overall vegetation distribution. Additionally, transferring endmembers in time assumes similar spectral vegetation properties throughout the time period studied. This could be problematic if structural and biophysical vegetation attributes changed, but field visits do not suggest this. It is important to note, though, that our study was not aiming for validating SMA fractions and that small differences in SMA vegetation fraction accuracy do not affect our conclusions about the trade-off involved when using qualitative measures.

Several implications for mapping gradual vegetation cover dynamics in drylands arise from our work: Spectral mixture analysis has so far been the method of choice for quantifying vegetation cover for subsequent time-analyses in drylands, mainly because SMA has been shown to outperform qualitative vegetation measures. Yet, SMA requires a large amount of user input and is not readily applicable to large areas. Our analyses shows that the trade-off involved when relying on simple vegetation indices may be small (i.e., in the order of sensor noise) for most applications. Time series analyses based on qualitative measures appear to reliably capture gradual vegetation cover change due to land use modifications. Qualitative measures have the additional advantage that they may require only simple relative radiometric normalization and easily allow for applying multi-sensor time series (e.g., Landsat TM and MSS). Moreover, qualitative estimates worked well in a highly heterogeneous Mediterranean landscape where the advantages of SMA should be strongest. Differences in vegetation estimates, though small in our study, may still be important for quantifying critical ecosystem services in drylands, e.g. freshwater provision or soil erosion protection. While differences among vegetation measures were small for areas where vegetation cover changed gradually, some qualitative measures (e.g., NDVI) performed poorly compared to SMA when capturing disturbance events such as fire. Likewise, we caution against generalizing our results to areas where non-linear relationships between vegetation estimates and vegetation cover are observed (e.g., in areas of dense vegetation). Overall, our study showed that trend analyses based on qualitative measures bear great promise for mapping gradual vegetation change in drylands across large areas. Thus, they can ultimately support to reconstruct vegetation dynamics and land use histories and to assess climate change impacts on these ecosystems.

Acknowledgments

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Chapter III:
Analyzing dense Landsat time series to assess the relationship between fire and grazing on Crete (Greece)

Remote Sensing of Environment (submitted)

Ruth Sonnenschein, Tobias Kuemmerle, Robert E. Kennedy, Warren B. Cohen, and Patrick Hostert

Abstract

In the Mediterranean Basin, fire frequency and extent have increased substantially during the last decades, with both climate and land use changes as potential drivers of fire regimes. Sheep and goat grazing is the primary land use in Mediterranean rangelands and closely linked to the fire as shepherds set fires and grazing controls fuel availability and connectivity. Here, we assess the link between grazing and fire regimes by analyzing gradual and disturbance-type vegetation cover change for the entire island of Crete (Greece). We used a dense time series of Landsat images from 1984 to 2010 and the trajectory change detection approach LandTrendr (Landsat based Detection of Trends in Disturbance and Recovery) to map fires and post-fire recovery (based on the Normalized Burn Ratio) and to quantify gradual vegetation trends (using Tasseled Cap Greenness). We found complex spatial patterns of vegetation changes, both in terms of fires and gradual trends. Fires affected a total area of 18,318 ha and vegetation on burnt patches needed on average 11 years to recover. The majority of fires were small and fire frequency decreased slightly throughout our observation period. Yearly climate variations appeared weakly related to fire dynamics. In contrast, changing grazing regimes and vegetation cover substantially influenced fire patterns and frequency. Pastoral fires were frequent in the foothills areas, whereas increasing vegetation cover in the mountains suggests a reduction of grazing pressure. Overall, the abandonment of traditional grazing practices like transhumance, increasingly leads to fuel accumulation and a higher risk of large and severe fires. This suggests that fire regimes in Crete's mountainous rangelands, which have traditionally been driven by land use, may increasingly be driven by climate in the future as a fuel limited fire regime is replaced by a drought driven fire regime. While this is worrisome in light of ongoing climate change, the upcoming Common Agriculture Policy (CAP) reform in 2013 represents a unique opportunity to counteract ongoing transformations of grazing systems (and thus fire regimes). Remote sensing, particularly the rich Landsat archives, play a crucial role in detecting transient and disturbance-type vegetation changes, even in heterogeneous environments such as Crete's rangelands. This provides new insights into the linkages between fire and land use, and the coupled effect of these drivers on vegetation patterns.

1 Introduction

Mediterranean ecosystems cover only 2% of the Earth's land surface (Medail and Quezel 1997), yet they provide ecosystem services to sustain the livelihoods of over 250 million people (Cox and Underwood 2011). These ecosystems are also a global biodiversity hotspot (Myers et al. 2000), containing about 20% of the world's vascular plants, many of which are endemic (Cowling et al. 1996). Due to the fire-prone climate with dry and hot summers, fire is a major agent shaping Mediterranean ecosystems, and many plants are specifically adapted to fire (Keeley et al. 2011). Fires in Mediterranean regions also threaten human life, erode ecosystem services, and may contribute to land degradation (Lampin-Maillet et al. 2010; Shakesby 2011). Better understanding of the patterns and underlying drivers of fire regimes as well as post-fire recovery in Mediterranean ecosystems is therefore a research priority.

While natural ignitions of fires occur, for example via lightning, anthropogenic factors are the main drivers of fire frequencies and patterns in many Mediterranean regions (Syphard et al. 2009). In the Mediterranean Basin, among the regions with the longest land use histories worldwide (Klein Goldewijk et al. 2010), fire has frequently been employed by humans for agro-pastoral activities and has shaped highly heterogeneous landscapes since prehistoric times (Naveh 1975b). However, for most European Mediterranean countries, both the number of fires and the total area burnt have increased exponentially since the 1960s (Moreno et al. 1998; Pausas and Vallejo 1999) and large fires have become more frequent (Moreira et al. 2011). These trends may at least in part be related to climate, for example, increasing occurrence of hot and dry summers, or precipitation variability that has caused severe droughts, which in turn favors large fires. However, the vast majority of ignitions are manmade and many of these fires appear to be related to land use changes (Pausas 2004).

The nexus between land use and fires is twofold. First, pastoral fires are set by shepherds to suppress the development of unpalatable woody vegetation and to improve forage quality of Mediterranean ecosystems, especially if they are used for livestock grazing (Papanastasis 2004). Second, increasing fire frequency and severity are also connected to an ongoing and accelerating trend of rural depopulation and abandonment of traditional land use practices (e.g., transhumance and terracing), due to urbanization. The accession of the northern Mediterranean countries to the European Union in the early 1980s amplified this twofold process of intensification on productive lands and rural exodus on marginal lands (Caraveli 2000), because the Common Agricultural Policy (CAP) provided strong financial incentives to intensify production (e.g. per-head livestock payments), thereby contributing to the abandonment of mountain pastoralism (MacDonald et al. 2000). Abandonment leads to fuel accumulation in the landscape and replaces the traditional, fine-scale land uses mosaic with continuous fuel beds (Moreira et al. 2001; Pausas 2004). Although the CAP was reformed to counteract the loss of traditional farming landscapes (Caraveli 2000), overgrazing and undergrazing of rangelands has been reported for many places. Fires are set more frequently to satisfy livestock feed demands (Xanthopolous 2000; Papanastasis 2004) thereby decreasing ecosystem resilience (Diaz-Delgado et al. 2002). Moreover, many Mediterranean countries are undergoing rapid expansion of tourism facilities, often close to wildlands, and as a result fires often put people and infrastructure at risk (Lampin-Maillet et al. 2010).

Despite the links between land use and fire in the Mediterranean Basin, the importance of land use relative to climate factors in driving fire regimes in the Mediterranean remains unclear, which is worrisome in light of rapid transformation of land use systems and accelerating climate change. Especially the influence of livestock husbandry on fire frequencies and spatial patterns in the Mediterranean Basin remains weakly understood. One important question in this context is whether fires are predominately associated with declining vegetation (overgrazing hypothesis) or increasing vegetation (rural exodus hypothesis) prior to ignition. A better understanding of the relative importance of land use related drivers, and particularly the effects of livestock on fire regime is urgent and timely considering the upcoming CAP reform, due in 2013.

One reason for the existing knowledge gap regarding the relationship between fire and land use intensity is that temporally and spatially detailed information on vegetation trends and fire patterns are missing for most places in the Mediterranean. Remote sensing time series analyzes can provide such information and map both abrupt and gradual changes in vegetation cover (Verbesselt et al. 2010). The Landsat data archive is particularly valuable for mapping vegetation trends in heterogeneous Mediterranean landscapes, because these sensors have been collecting fine-scale images since the early 1980s. However, existing studies in Mediterranean ecosystems have so far mainly focused on either grazing-induced vegetation changes (Hostert et al. 2003a; Röder et al. 2008b), fire patterns (Diaz-Delgado and Pons 2001; Röder et al. 2008a; Bastarrika et al. 2011) or post-fire recovery (Viedma et al. 1997; Stow et al. 2007; Veraverbeke et al. 2010). Assessments of the relationship of both, gradual and abrupt vegetation changes in the Mediterranean are missing, despite the potential of such studies to better understand the link between grazing and fire regimes.

The opening of the USGS Landsat archives provides new and exciting opportunities to reconstruct detailed land use and land cover histories (Landsat Science Team et al. 2008; Wulder et al. 2008). However, change detection approaches that focus simultaneously on rapid, disturbance-type and gradual change processes are scarce. Trajectory-based Landsat time series analysis allows for the semi-automatic characterization of long-term and short-term change (Kennedy et al. 2007; Huang et al. 2010; Kennedy et al. 2010), thus enhancing traditional time series interpretation approaches. These algorithms have been developed for forest ecosystems, and adapting them to Mediterranean-type ecosystems would provide new avenues to assess subtle vegetation changes and fire disturbances.

Our overall goal was to better understand the relationship of land use and fire regimes in Mediterranean rangelands by mapping both rapid and gradual vegetation cover change using a trajectory change detection approach. We focus on Greece's largest island Crete, because it spans a range of climate and topography conditions, and entails a large proportion of semi-natural rangelands. Animal husbandry is the most widespread land use on Crete, fires are frequent, and grazing systems have been transformed substantially following the country's accession to the European Union in 1981 (Dubost 1998). Together, this renders Crete an interesting study region to analyze the temporal and spatial effects of grazing on fire regimes. Our specific research questions were:

1. How did vegetation cover change in Crete's rangelands during the period 1984-2010 in terms of gradual vegetation trends, fires, and post-fire recovery?
2. What is the relationship between fire frequency and extent and variations in climate, topography, and grazing pressure?

2 Study region

Crete, the fifth-largest island (8,265 km²) in the Mediterranean Sea, is located about 160 km south of the Greek mainland (Fig. III-1). The elongated island spans 260 km from east to west and 60 km at its widest point from north to south (NSSG 2009). Terrain is rugged with four east-west mountain ranges (White Mountains, Psiloritis, Dikti and Thripti), and elevations up to 2,456 m. Bedrock is heterogeneous, with crystalline and platy limestone forming the massifs, phyllites, quartzites and younger flysch deposits in the lower mountains, and marls, sandstone, and clays in the plains (Rackham and Moody 1996). Climate is temperate-mediterranean, with hot and dry summers, mild and wet winters, and strong elevational gradients. Average temperatures are 26° C in summer and 12° C in winter at sea level (Hellenic National Meteorological Service 2011). Rainfall dominantly occurs from October to March, but varies greatly across the island and among years, with mean precipitation from 300 mm in the southeast to 700 mm in the northwest, and up to 2,000 mm in the mountains (Chartzoulakis et al. 2001). Strong local winds prevail the entire year.

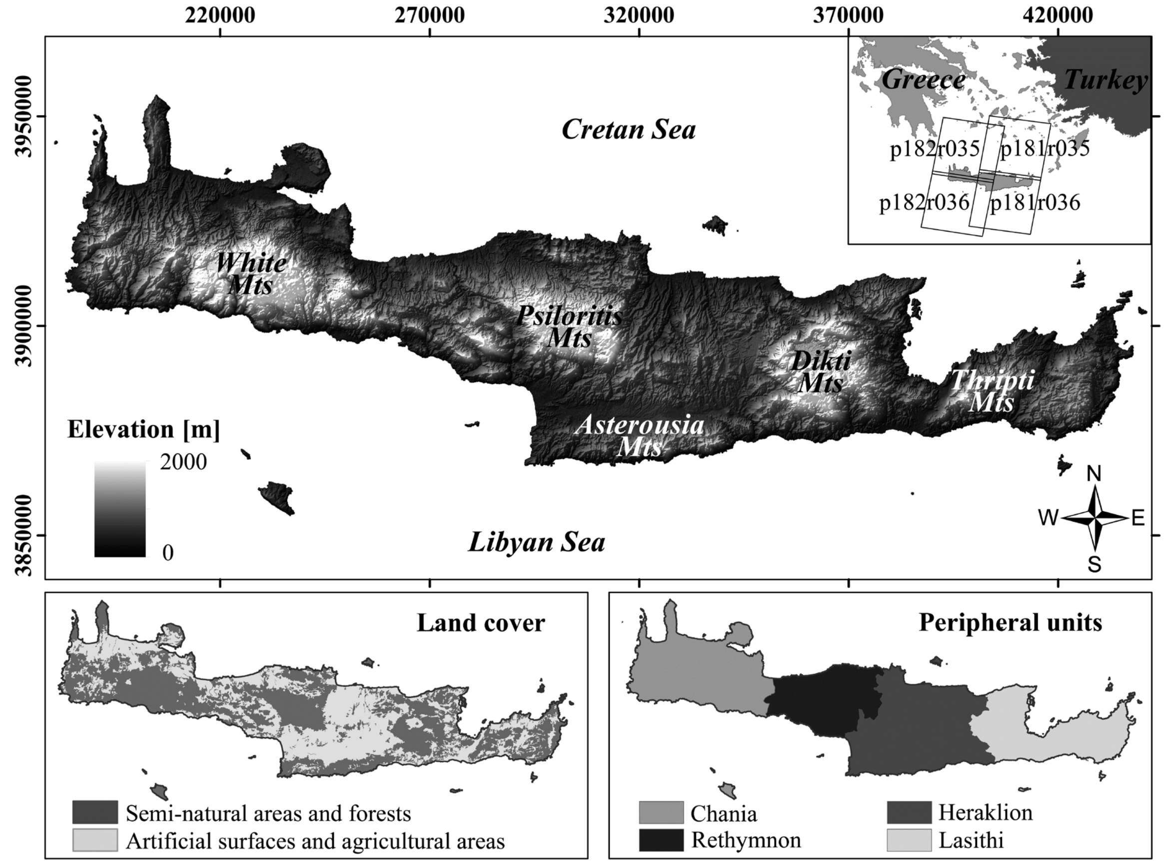

Figure III-1: The study area of Crete shown as a relief representation, its location within the Mediterranean Sea and its coverage by the four Landsat footprints. The extents of rangelands on Crete are shown as well as the administrative boundaries of the peripheral units.

The vegetation of Crete is typical of the eastern Mediterranean, adapted to dry climate. Due to the long land use history, natural vegetation has been largely transformed and native forest (*Pinus brutia, Cypressus sempervirens, Quercus coccifera)* persist only on the flanks of the mountain ranges. Phrygana (seasonally dimorphic dwarf shrubs, e.g. *Sarcopoterium spinosum, Phlomis fruticosa*) is the most dominant vegetation type, followed by matoral (woodland degraded to shrubs), while grasslands are limited to dolines and flat areas with relatively deep soils (Rackham and Moody 1996).

Crete is one of the 13 regions (i.e., states) of Greece, subdivided into four peripheral units (i.e., the former prefectures Chania, Rethymno, Heraklion, Lasithi) (Fig. 1). Population increased from ~502,000 inhabitants in 1981 to ~601,000 inhabitants in 2001, with 58% living in urban areas (NSSG 2009). Agriculture is an important source of income for rural populations. The plains are intensively farmed, mainly for olives (the main agrarian export product), grapes, vegetables, and grain. Farming has intensified since the 1980s (e.g. expansion of irrigation agriculture and greenhouses), sometimes resulting in a severe depletion of the groundwater table (Croke et al. 2000).

Sheep and goat grazing in rangelands is the most widespread land use activity on Crete, covering about 50% of the island (Fig. III-1). While rangelands at lower elevations are grazed during the entire year, mountain pastures were traditionally only grazed during summer. Rangelands are private or managed under communal arrangements. Grazing is a legal right in all rangelands, but conversion to other land uses is prohibited, as are fires in forests and in their surroundings during the summer and burnt areas must be excluded from grazing for up to 10 years to allow for recovery (Papanastasis 1993). With Greece's EU accession in 1981, animal numbers increased from ~713,600 to 1,878,900 (sheep) and 329,100 to 654,900 (goats) until 2008 (NSSG 2009). Stocking rates increased and exceed carrying capacity (Papanastasis 1998) and overgrazing and frequent pastoral fires have been reported for many areas (Lyrintzis and Papanastasis 1995; Papanastasis et al. 2002). At the same time, traditional management practices like transhumance declined in importance, potentially leading to fuel accumulation and higher risk of wildfires (Rackham and Moody 1996).

3 Data

3.1 Satellite images

To map vegetation change on Crete, we acquired a dense time series of TM/ETM+ images. Because phenology in Mediterranean ecosystems varies between years, we selected peak vegetation Landsat images based on broad-scale, hypertemporal time series (Sonnenschein et al. 2011). For 1989 to 2005, we used the 10-day composites from the 'Mediterranean Extended Daily 1-km AVHRR Data Set' (MEDOKADS) (Koslowsky 1998). For 2000 to

2010, we used the 250-m MOD13Q1 vegetation index product (version 5). We reduced noise in the time series using a Savitzky-Golay filter and derived annual peak NDVI dates for each year and pixel based on a Fourier transformation. While phenology was variable in space, the timing of peak vegetation was consistent from year to year (Stellmes et al. 2010; Sonnenschein et al. 2011). We then selected Landsat TM/ETM+ images from 1984 to 2010 for the four footprints covering Crete (path/row: 181/035, 181/36, 182/35, and 182/036) acquired close to mountainous rangeland vegetation peak dates. Where clouds occurred, additional images were used to fill gaps (Table III-1).

Table III-1: Acquisition dates and sensors of the Landsat imagery used in this study.

Path/row	*Landsat images used for the change detection*	*Additional, Landsat images used to establish a fire reference data base*
181/036 and 181/035	1984/06/19^{TM5}, 1985/05/21^{TM5}, 1986/05/24^{TM5}, 1987/06/12^{TM5}, 1987/06/28^{TM5}, 1988/05/29^{TM5}, 1989/06/17^{TM5}, 1991/06/07^{TM5}, 1993/06/12^{TM5}, 1994/06/15^{TM5}, 1996/06/04^{TM5}, 1997/05/22^{TM5}, 1998/06/26^{TM5}, 2000/06/07^{ETM+}, 2001/06/10^{ETM+}, 2002/05/20^{TM5}, 2002/06/05^{TM5}, 2002/06/29^{ETM+}, 2003/06/24^{TM5}, 2004/06/10^{TM5}, 2004/06/26^{TM5}, 2005/05/20^{ETM+}, 2005/06/05^{ETM+}, 2005/06/13^{TM5}, 2006/05/15^{TM5}, 2006/05/23^{ETM+}, 2006/05/31^{TM5}, 2008/05/28^{ETM+}, 2008/06/21^{TM5}, 2009/05/23^{TM5}, 2009/05/31^{TM5}, 2009/06/16^{ETM+}, 2010/05/10^{TM5}	1984/10/09^{TM5}, 1985/09/26^{TM5}, 1986/08/28^{TM5}, 1987/09/16^{TM5}, 1990/08/07^{TM5}, 2000/09/11^{ETM+}, 2001/08/29^{ETM+}, 2002/10/19^{ETM+}, 2003/09/12^{TM5}, 2004/09/30^{TM5}, 2005/09/01^{TM5}, 2006/10/06^{TM5}, 2007/08/22^{TM5}, 2008/08/24^{TM5}, 2009/08/27^{TM5}, 2010/08/30^{TM5}
182/036 and 182/036	1984/06/10 ^{TM5}, 1985/06/29 ^{TM5}, 1986/05/31 ^{TM5}, 1987/05/18 ^{TM5}, 1987/06/03 ^{TM5}, 1988/05/28 ^{TM4}, 1989/06/24 ^{TM5}, 1991/06/14 ^{TM5}, 1993/06/19 ^{TM5}, 1994/05/21 ^{TM5}, 1996/06/27 ^{TM5}, 1997/06/30 ^{TM5}, 1998/06/17^{TM5}, 2000/05/13^{ETM+}, 2000/06/30^{ETM+}, 2001/05/16^{ETM+}, 2002/05/19^{ETM+}, 2002/06/12^{TM5}, 2002/06/28^{TM5}, 2003/05/06^{ETM+}, 2004/06/25^{ETM+}, 2005/06/28^{ETM+}, 2006/05/14^{ETM+}, 2006/05/30^{ETM+}, 2006/06/15^{ETM+}, 2007/05/17^{ETM+}, 2007/06/02^{ETM+}, 2007/06/18^{ETM+}, 2008/05/03^{ETM+}, 2009/05/06^{ETM+}, 2009/05/22^{ETM+}, 2010/05/09^{ETM+}, 2010/06/18 ^{TM5}	1984/09/30^{TM5}, 1985/06/29^{TM5}, 1986/08/19^{TM5}, 1987/09/23^{TM5}, 1990/08/14^{TM5}, 1999/08/15^{TM5}, 2000/09/02^{ETM+}, 2001/09/21^{ETM+}, 2002/08/07^{ETM+}, 2003/09/03^{TM5}, 2004/09/13^{ETM+}, 2005/08/15^{ETM+}, 2006/09/19^{ETM+}, 2007/07/20^{ETM+}, 2008/09/08^{ETM+}, 2009/08/18^{TM5}, 2010/07/20 ^{TM5}

In total, we used 61 images, resulting in a time series of at least one image every two years. For validation purposes, we also acquired all available cloud-free images from late summer and autumn to help establishing a reference fire database (33 additional images, Table III-1).

3.2 Other data

To stratify the study region, we obtained the CORINE Land Cover map from 2000 (http://www.eea.europa.eu/) at a scale of 1:100,000. Administrative boundaries of the peripheral units were available from the governmental service of Greece (http://geodata.gov.gr). Annual and seasonal precipitation and mean temperature from 1981 to 2010 were obtained from the E-OBS database (Tank et al. 2002) for the Heraklion climate station. We gathered annual livestock numbers for goats and sheep and information on animal keeping as well as areas of fodder plants for grazing for the four peripheral units from 1981 to 2006 (National Statistical Service of Greece 1982-2010). We also acquired the 90-m Shuttle Radar Topography Mission (SRTM) digital elevation model and resampled it to the Landsat resolution (http://www2.jpl.nasa.gov/srtm).

4 Methods

4.1 Landsat preprocessing

To reduce the number of images to handle, we only acquired images from the same acquisition date within paths and mosaicked them, yielding two Landsat time series (182/035-036 and 181/035-036, hereafter). Seventy-five images were provided by the United States Geological Survey (USGS) already ortho-rectified (LT1). The other 19 images were acquired from Eurimage and co-registered to the USGS images using a semi-automatic tie point algorithm that accounts for topographic displacement (Hill and Mehl 2003; Kuemmerle et al. 2006). Positional uncertainty was generally < 0.5 pixels (15 m). To radiometrically normalize our 61 images, we followed the protocol incorporated in LandTrendr (Landsat-based Detection of Trends in Disturbance and Recovery, Kennedy et al. (2010)), which calibrates all images to a reference image. We chose the image from 14 June 1991 as reference, because this image showed clear atmospheric conditions and was acquired close to the mid year and also to the median day of year of our time series. To ensure comparability across paths, we used the same reference image for both footprints (26% overlap between footprints). First, we applied the COST-model (Chavez 1996) to the reference image, which integrates radiometric calibration and a dark objection approach. Dark objects were chosen from shaded areas in steep topography and we assessed band-wise histograms to select the darkest pixels within these target object. Second, all other images were normalized to the reference image using the multivariate alteration detection and calibration algorithm (MADCAL, Canty et al. (2004). Radiometric consistency in our time series was high ($R^2 > 0.95$ within footprints, $R^2 > 0.99$ between adjacent footprints).

We Tasseled Cap transformed all Landsat images and screened them for clouds, cloud shadows, and snow using a reference image to contrast affected areas (Kennedy et al. 2010). We then manually determined thresholds to mask cloud, cloud shadow and snow, and where necessary, corrected errors in these masks. We also masked radiometric (saturation effects in red and near-infrared bands) and geometric distortions (spatial shift of scan line errors).

4.2 Trajectory-based change analysis

We applied the LandTrendr trajectory-based change detection (Kennedy et al. 2010) to identify both abrupt und gradual vegetation changes. LandTrendr assesses temporal profiles of spectral indices by fitting linear segments on a per pixel basis, using regression methods and point-to-point fitting, to identify periods of stability, increase, or decrease. The sensitivity of LandTrendr to detect these changes can be influenced by different parameter settings controlling the number and location of segments (Table III-2). The final set of segments was determined automatically using a statistical goodness-of-fit criterion (the F-value) (Kennedy et al. 2010). Masks were used to flag missing pixels in the temporal profiles and information from additional images from the same year was used to fill gaps (using the image closest to the peak vegetation date).

In a previous study, we showed that patterns of gradual changes in drylands are captured equally well by Tasseled Cap Greenness (TC_G) and Spectral Mixture Analysis (SMA) (Sonnenschein et al. 2011) and we therefore chose the TC_G as our index for LandTrendr. Because initial tests showed that vegetation-based indices can overlook fires that may have occurred in the previous year (after image acquisition of the summer image), we also carried out a LandTrendr analysis using the Normalized Burn Ratio (NBR) which combines near-infrared ($\rho 4$) and mid-infrared ($\rho 7$) reflectance (Key and Benson 2003) to capture fires:

$$NBR = \frac{(\rho 4 - \rho 7)}{(\rho 4 - \rho 7)} \qquad (1)$$

We tested different combinations of segmentation parameters (Table III-2) based on a reference database of 50 fire-affected areas identified in the late autumn images for different years across the study region as well as 50 areas where vegetation had changed gradually based on our previous study (Sonnenschein et al. 2011). We visually compared the fitting of linear segments of our reference database and chose the parameterization for each index that captured the different change process best for our study region (Table III-2).

LandTrendr generates a wealth of information because temporal segmentation describes the entire change dynamic for a given pixel over the entire observation period (Kennedy et al. 2010). To summarize this information, LandTrendr labels individual segments or sequences of segments according to a user-defined ruleset consisting of four indicators to describe abrupt and gradual index (i.e., vegetation) change per pixel: (1) onset of change event, (2) relative magnitude, (3) duration, and (4) index value prior to the change event. Where subsequent change processes are detected (e.g. fire and recovery), separate indicator sets are derived for each event. Moreover, several parameters can be set to reduce false detections by restricting vegetation cover prior to disturbances, defining yearly and long-term change magnitudes and collapsing temporally adjacent segments describing the same change process.

Table III-2: Parameterization of the LandTrendr segmentation.

Parameter	*TC_G*	*NBR*
Number of segments	4	6
f-value	0.5	0.5
Best model proportion	0.9	0.9
Distweight factor	2	2
Vertexcount overshoot	3	3
Desawtooth (despike)	0.9	1 (not used)
Recovery_threshold	1	1

Because in LandTrendr these parameters are set relative to vegetation cover, a model relating index values (TC_G and NBR in our case) to vegetation cover is needed. We used a cover model from our own previous work (Sonnenschein et al. 2011), where fractional vegetation cover had been derived from a Spectral Mixture Analysis (SMA) for Central Crete. To transfer the SMA cover model, we calculated linear regressions between SMA values and index values (R^2 = 0.73 and 0.72 for TC_G and NBR, respectively) and transformed TC_G and NBR values to percentage vegetation cover.

We labeled the following change classes: fire disturbance event, long-term gradual increase, and long-term gradual decrease (Table III-3). To do so, we merged subsequent segments fitted to the time series of index values which had an angle < 15° between them. To minimize confusion due to phenology, we set a minimum of 15% absolute and 30% relative vegetation cover change for fire pixel. Likewise, we only considered long-term gradual trends of at least 20 years and used a threshold of 5% cover change to consider an area changed, according to Hostert et al. (2003a). In the same way, we also captured the vegetation trends of pre-disturbance (fire) segments (i.e., increase and decrease) and the post-fire recovery process as well as the occurrence of repeated fires (Table III-3). To further assess post-fire recovery rates, we derived the time required for each disturbed pixel to recover to 50% of its pre-disturbance vegetation cover and to recover completely.

Table III-3: Labeling of change classes.

Label of change class	*Index*	*Segments*	*Definition*
Fire disturbance	NBR	1	Duration of segment < 3 years with disturbance magnitude > 20% vegetation cover
Long growth	TC_G	1	Increase in segment for > 20 years and > 5% vegetation cover
Long decline	TC_G	1	Decrease in segment for > 20 years and < 5% vegetation cover
Repeated fire disturbances	NBR	2	Duration of first segment < 3 years with disturbance magnitude > 20% and duration of second segment < 3 years with disturbance magnitude > 20%
Fire disturbance and recovery	NBR	2	Duration of first segment < 3 years with disturbance magnitude > 20% and increase of second segment
Growth before fire	NBR	2	Increase in first segment by an annual rate of > 0.25% and duration of second segment < 3 years with a disturbance
Decline before fire	NBR	2	Decrease in first segment by an annual rate of > 0.25% and duration of second segment < 3 years with a disturbance

We mosaicked the resulting maps for both paths, using the path with the highest number of images (181/035-036) for the overlap area. Because our focus here was on rangelands, we masked all other land covers based on the CORINE map (classes forest and semi-natural areas) and excluded areas > 2,000 m and areas > 1,600 m for the White Mountains, where frequent snow cover hindered the change detection. Finally, we used a 1-ha minimum mapping unit (11 pixels) and removed all fire patches smaller than this threshold. We also summarized fire patches for the size classes of 1-5 ha, 5-25 ha, 25-50 ha and > 50 ha.

4.3 Validation

To validate the LandTrendr results, we used TimeSync, a tool to visualize and interpret time series of Landsat image stacks (Cohen et al. 2010). For a given pixel, TimeSync displays the Landsat time series as chronological sequences of image chips and spectral index values through time, together allowing interpreters to label trajectories (Cohen et al. 2010). We gathered a reference dataset based on a stratified random sample of points. Following Schroeder et al. (2011), strata were defined by the onset year of fires. We gathered 30 points per fire disturbance year (21 years) as well as 200 points for long-term gradual changes (increase and decrease) and no-change resulting in a total of 830 samples. We interpreted all temporal profiles with TimeSync by simultaneously assessing image chips and temporal profiles of TC_G and NBR. We also used Landsat images from late summer to detect fire disturbances and high-resolution imagery in Google Earth to identify recent conversions of rangelands. We then compared the reference database against the LandTrendr change results, calculated an error matrix to validate fire onset and derived overall accuracy as well as omission and commission errors. To account for the area proportions for both error matrices, we calculated area-adjusted accuracy measures (Card 1982) and true area-estimates for all classes as well as their 95% confidence intervals (Cochran 1977).

4.4 Temporal and spatial patterns of vegetation changes

To analyze the relationship of climate and fire, we compared time series of fire occurrences and fire extents with time series of annual and summer (June, July, August) precipitation and temperature throughout our observation period (1981 - 2010). Most fires occurred in late summer, after the acquisition of our Landsat imagery, and were thus registered by LandTrendr only in the following year. To account for this, we compared fire occurrences to the climate variables of the previous year (i.e., shifting the fire time series by one year backwards) and calculated Pearson’s correlation coefficient (R) as well as tested the significance of R. To also assess the impact of precipitation on fuel loads (Pausas 2004), we additionally shifted the fire occurrence time series two and three years backwards.

Similarly, we summarized fire occurrences and extents for four elevation classes (0 – 400 m, 400 – 800 m, 800 – 1200 m, 1200 – 2000 m). We also compared absolute vegetation cover and relative cover changes prior to fire disturbances as well as the post-fire recovery rates among these elevation classes. Likewise, we summarized gradual vegetation changes for all elevation levels.

To analyze differences in fire characteristics and gradual vegetation trends between the four peripheral units, we summarized fire occurrences, extent, and areas of gradual vegetation increase and decrease for each of the four administrative units. To assess the relationship of vegetation cover change, both concerning fire and concerning gradual change, and livestock husbandry, we calculated livestock stocking rates by dividing the total number of sheep and goats for each year through the rangeland area of each peripheral unit and compared stocking rates as well as crop areas of fodder plants for grazing (barley, oats, vetch, etc) to the vegetation trends and fire frequencies and extent.

5 Results

We found widespread vegetation cover changes across the island of Crete, both in terms of gradual trends and abrupt disturbances (Fig. III-2). Overall, 17% (~76,099 ha) of all rangelands were affected by a vegetation cover change process (fire, gradual decrease or increase) during our observation period. Concerning fires, we detected a total of 5,174 fire events that burned a cumulative area of ~18,318 ha. Repeated burning was almost absent and we did not find widespread evidence that abrupt vegetation disturbances were due to other causes than fire (we saw the expansion of olives into rangelands in some areas, e.g., in Chania, but that was not extensive). Fires occurred across a wide range of pre-fire vegetation cover levels, with a mean vegetation cover value of 48% and a standard deviation of 14%. Concerning vegetation trends prior to fire events, 13% of all pixels within fire patches showed a vegetation cover decrease before the actual fire event, while 55% of all fire pixels were characterized by a vegetation cover increase. Following a fire, 92% of all pixels showed recovery, but only about 13% recovered entirely. Average recovery time to pre-fire conditions was 11 years, variability in recovery time, however was large. Recovery time had a standard deviation of 5.5 years, with some patches recovering rapidly (within 2 years), and very slow or no recovery on others (e.g., where rangelands were converted to agricultural fields following a fire). Post-fire recovery to 50% of pre-fire vegetation cover, had a mean value of about 4 years.

Gradual vegetation changes were also widespread on Crete, affecting a total of 13% of the island. Gradual changes split nearly equally in increasing and decreasing vegetation trends (6% and 7%) with overall mean changes of 9% and 8% vegetation cover for increasing and decreasing trends, respectively. Initial vegetation cover did not vary much between increasing and decreasing trends in rangeland vegetation with mean values of 32% vegetation cover and 29%, respectively.

Landtrendr yielded a highly accurate fire disturbance map with an overall accuracy of 95.5% (Table III-4). Omission errors for the no-change class was 0% while those for single disturbance years were generally moderate (mean = 29.6%, standard deviation = 33.2%) and varied strongly among years (Table III-5). Highest values generally occurred in the beginning of the observation period. Commission errors were generally lower than the omission errors (mean = 19.0%, standard deviation = 13.6%) with low values in 1991 and highest values in 2005 and 2010. Both errors decreased for single years (mean omission error = 29.3% and mean commission error = 14.3%) when including all disturbance events (i.e. conversions) that result in a similar breakdown in the temporal profile of NBR while the overall accuracy remained similar (95.6%).

Table III-4: Error matrix of fire disturbances.

Landtrendr \ Reference	1985	1986	1987	1988	1989	1991	1993	1994	1996	1997	1998	2000	2001	2002	2003	2004	2005	2006	2008	2009	2010	no change	total
1985	24	4																				2	30
1986		25	1	1																		3	30
1987			25	3																		2	30
1988				27	1																	2	30
1989					29																	1	30
1991						30																0	30
1993							26															4	30
1994								24	5													1	30
1996								1	27													2	30
1997										23												7	30
1998				1							24	2										3	30
2000												29										1	30
2001													24	3	1							2	30
2002														27								3	30
2003															22	1						7	30
2004																24	2					4	30
2005																2	15	8				5	30
2006																		24	2			4	30
2008																			27			3	30
2009																				20		10	30
2010		1					2													3	14	10	30
no change		1	1		1	2		1	1			1										192	200
total	24	31	27	32	31	32	28	26	33	23	24	32	24	30	23	27	17	32	29	23	14	268	830

Table III-5: Results of the area-adjusted accuracy assessment of the disturbance map showing omission and commission errors for each year and overall accuracy (OA). The accuracy for fire disturbances are shown in the second and third column while those of all detected disturbances (including conversions) are shown in the last two columns.

	Fires		*Disturbances*	
Year	*Omission*	*Commission*	*Omission*	*Commission*
1985	0%	20%	0%	20%
1986	84%	17%	83%	13%
1987	85%	17%	86%	17%
1988	6%	10%	6%	10%
1989	69%	3%	69%	0%
1991	87%	0%	88%	0%
1993	5%	13%	4%	3%
1994	71%	20%	71%	17%
1996	61%	10%	60%	3%
1997	0%	23%	0%	7%
1998	0%	20%	0%	17%
2000	62%	3%	62%	0%
2001	0%	20%	0%	20%
2002	14%	10%	14%	7%
2003	3%	27%	2%	20%
2004	12%	20%	12%	17%
2005	13%	50%	12%	43%
2006	15%	20%	14%	13%
2008	7%	10%	7%	0%
2009	27%	33%	26%	27%
2010	0%	53%	0%	47%
No Change	0%	4%	0%	4%
OA = 95.5% (Fires)				
OA = 95.6% (Disturbances)				

The spatial pattern of fires and gradual vegetation change varied substantially across the island (Fig. III-2). Fires were frequent in Western Crete and in the north of the peripheral unit Rethymno. These areas were also often characterized by decreasing or increasing vegetation cover. In contrast, only a few fires occurred in the Asterousia Mountains where areas of strong vegetation decrease prevailed. Relatively large areas of increasing vegetation were found in the Psiloritis Mountains, and areas of decreasing vegetation cover were scarce there. Large-scale fires were abundant in the eastern part of Crete, where fire also sometimes spread into farming areas. In Eastern Crete, rangeland vegetation was also often characterized by long-term vegetation cover decline (Fig. III-2).

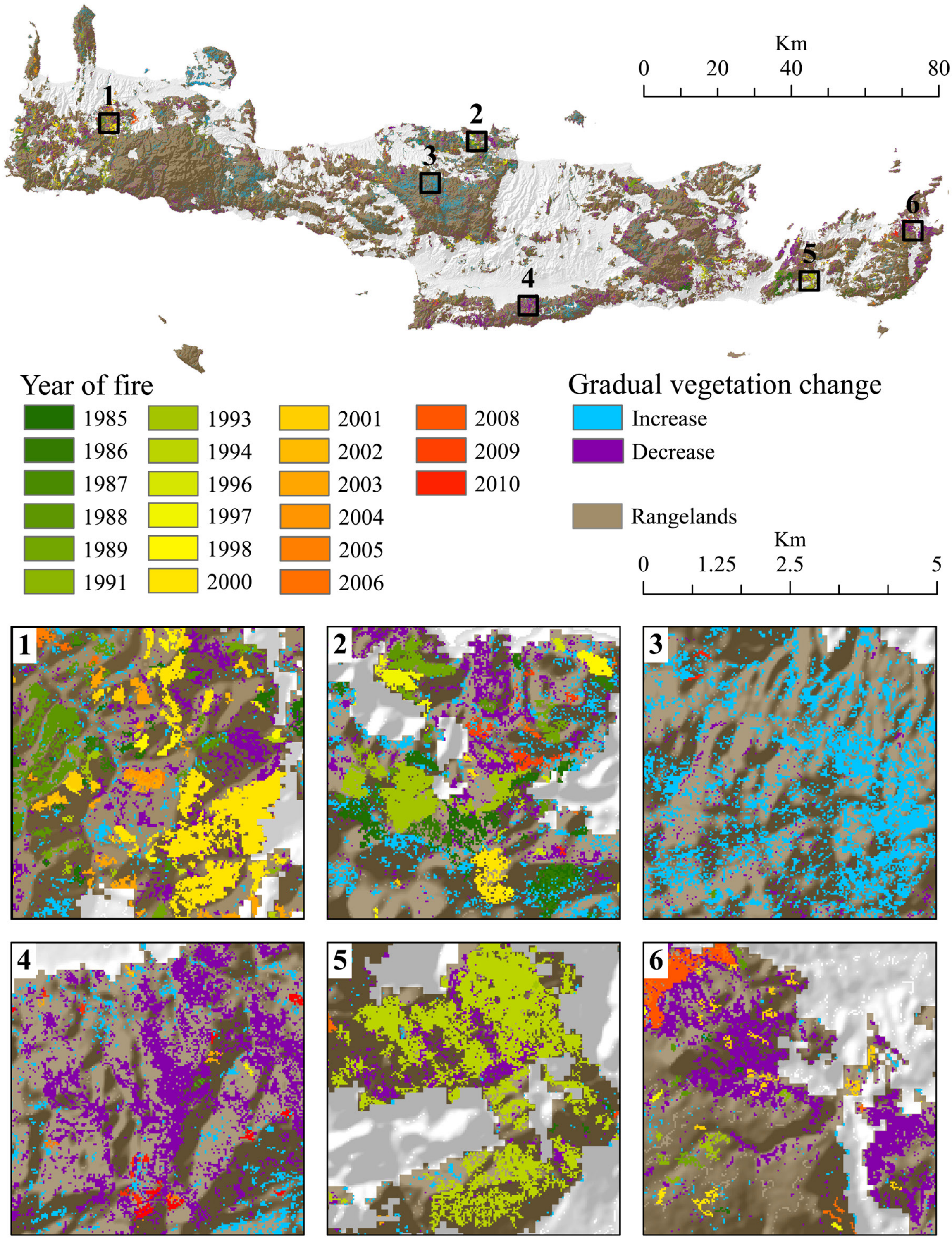

Figure III-2: Fire occurrences and gradual vegetation changes on Crete during the observation period 1984-2010.

The spatial pattern of pre-fire vegetation changes was complex and increasing or decreasing changes were often present within the same fire patch (Fig. III-3). Likewise, recovery time varied substantially within and among burnt areas.

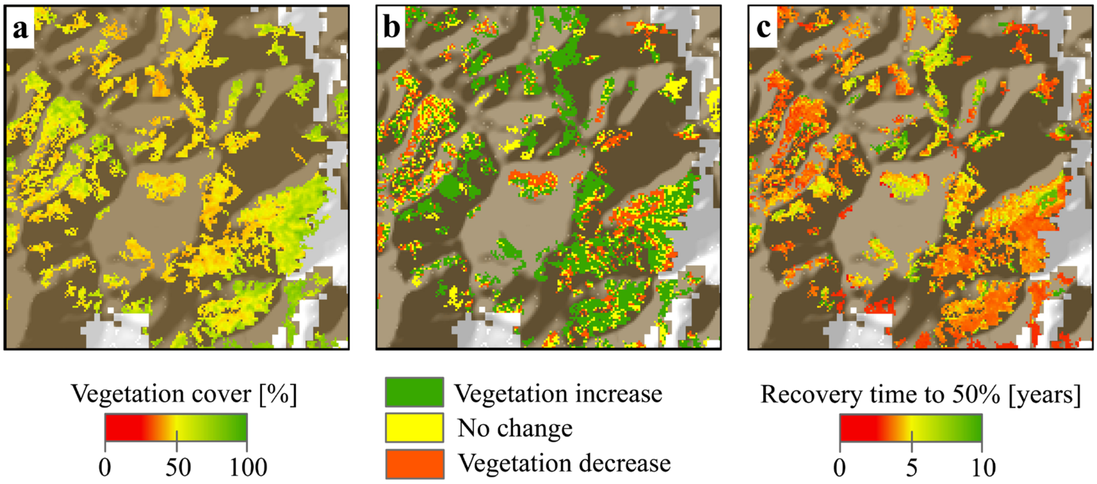

Figure III-3: Vegetation cover (a) and vegetation changes prior to fire events (b) as well as the subsequent recovery process (c). The area refers to the example 1 shown in Figure 2.

The number of fires and the area extent burnt varied substantially among fire patch size classes. Most fires (87%) were small, covering < 5 ha while accounting for 40% of the total burnt rangeland vegetation (Fig. III-4). The few large fires (31) accounted for a substantial part of to the total burnt area of Crete's rangelands (19%), with the largest fire burning an area of 262 ha (in 1985). Fire frequency decreased gradually with increasing fire size, while the area burnt decreased only for fires covering < 50 ha.

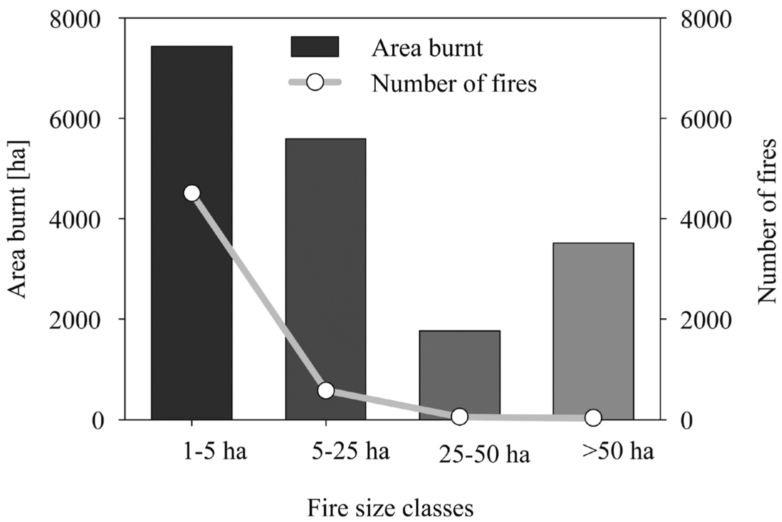

Figure III-4: Fire size distribution for 175ha, 5-25ha, 25-50ha and > 50ha.

Although fires were abundant in all years, the number of fires and the area burnt differed markedly between years (Fig. III-5). The highest total number of fires (777) occurred in 1985, while we found only 124 fires in 2002. Fires < 25 ha occurred in all years we studied, while fires larger this threshold were mostly missing in the less fire affected years. Large fires occurred especially in 1985 and 1996 and accounted for 37% and 32% of the total burnt area of these years, respectively. The annual number of fires decreased slightly after 1985, mainly driven by a decrease in medium and large fires (> 25 ha), with a similar trend for the overall area burnt. In contrary, mean fire size declined throughout our observation period, partly the result of the lack of large fires in most years of the recent decade.

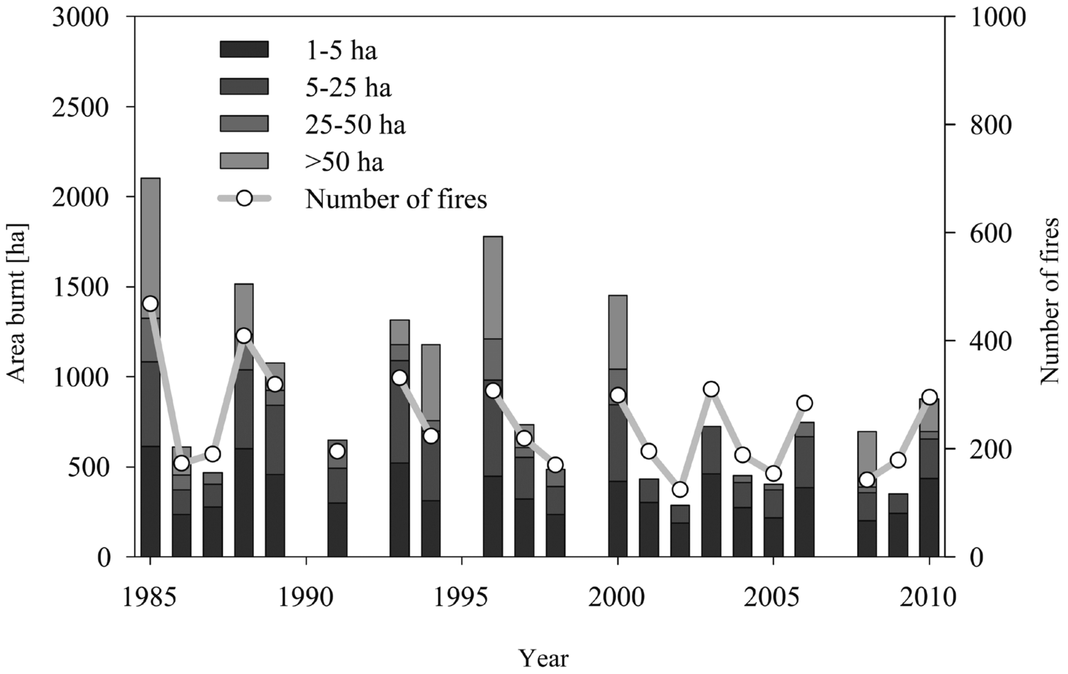

Figure 5: III-Fire frequency and area burned in 1985-2010.

Fire occurrences and the total burnt area did not show a clear relationship to climate in Crete. Temperature generally increased in 1981-2010 (Fig. III-6), and this increasing trend was more pronounced for mean annual temperature than for mean summer temperatures (which also showed a higher yearly variability). Precipitation declined throughout our observation period while annual precipitation amount varied between 282 mm in 2008 and 649 mm in 2001 (Fig. III-6). Summer precipitation appeared not related to annual rainfall patterns and varied substantially among years. Several consecutive years showed no or only marginal rainfall (e.g. 1985-1991 and 2003-2009), while other years had summer rainfall (e.g., exceptionally wet years from 1982-1984 and 2002). Variability of fire frequency and areas burnt for dry summers with only marginal rainfall was large and the years with summer rainfall did not show a strong correlation to the fire regime (Table III-6).

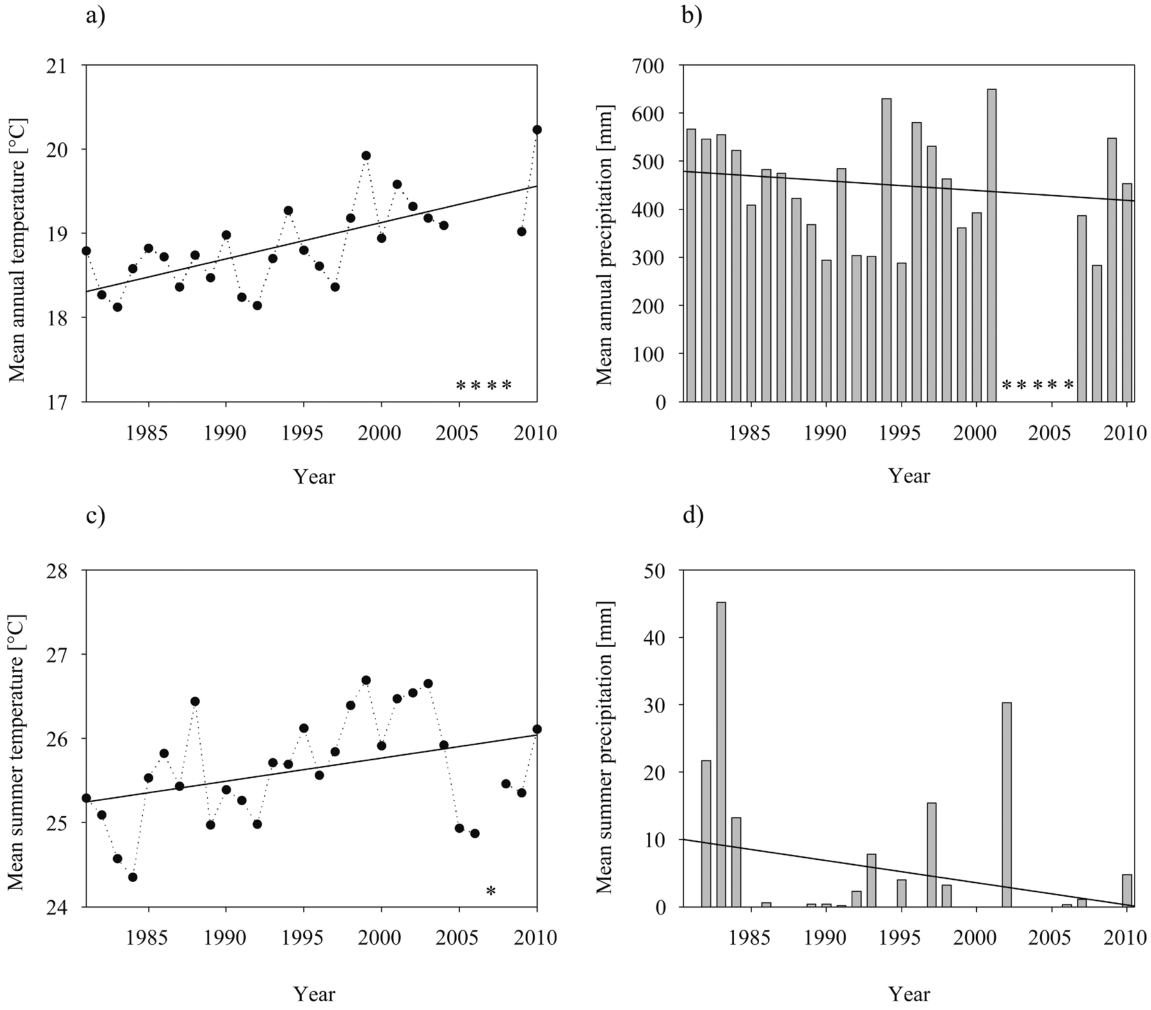

Figure III-6: Changes in mean annual and summer temperature (a and c) and in annual and summer precipitation (b and d) for the period 1981-2010. Missing data are marked with a (5).

Likewise, annual precipitation pattern generally explained variations in fire parameters only to some extent. Highest correlations were found between fire frequency and annual area burned and the year before the actual fire event (Table III-6).

Fire regime characteristics varied also substantially with topography. Lower elevations (0 – 800 m) showed markedly higher numbers of fires and total burnt area than higher elevations (Fig. III-7). Fires were present in all years in the lowlands although fire frequency varied strongly among years. In contrast, in the mountains most of the burnt area was from fire events from only a few years. Fire pre-disturbance values peaked for 400 – 800 m and generally decreased for elevations greater than 800 m, going along with an increasing percentage of pre-disturbance segments showing a decreasing vegetation cover trend. Recovery times also differed among elevation zones and fire patches in the mountains required most time to recover to 50% of their pre-disturbance vegetation cover. Fire-affected areas in elevations between 800 – 1200 m showed the highest recovery times on average while areas above this elevation showed the highest variability.

Table III-6: Correlations between fire regime and climate parameters.

	Annual rainfall (n = 17)	*Summer rainfall (n = 21)*
Fire frequency		
1-year shift	R = 0.02	R = 0.27
2-year shift	R = 0.38	R = 0.35
3-year shift	R = 0.08	R = 0.05
Burnt area		
1-year shift	R = 0.19	R = 0.16
2-year shift	R = 0.37	R = 0.35
3-year shift	R = 0.10	R = 0.07

This was different for gradual vegetation cover changes. Although their pre-disturbance values were similar to the fire pre-disturbance values, the percentage of rangelands where vegetation gradually declined decreased with elevation, while large proportions of the rangelands above 800 m showed increasing vegetation trends (Fig. III-7). Average absolute changes due to gradual increase did not show large variations while those due to gradual vegetation decrease declined slightly with increasing elevation (8% to 6% vegetation cover change).

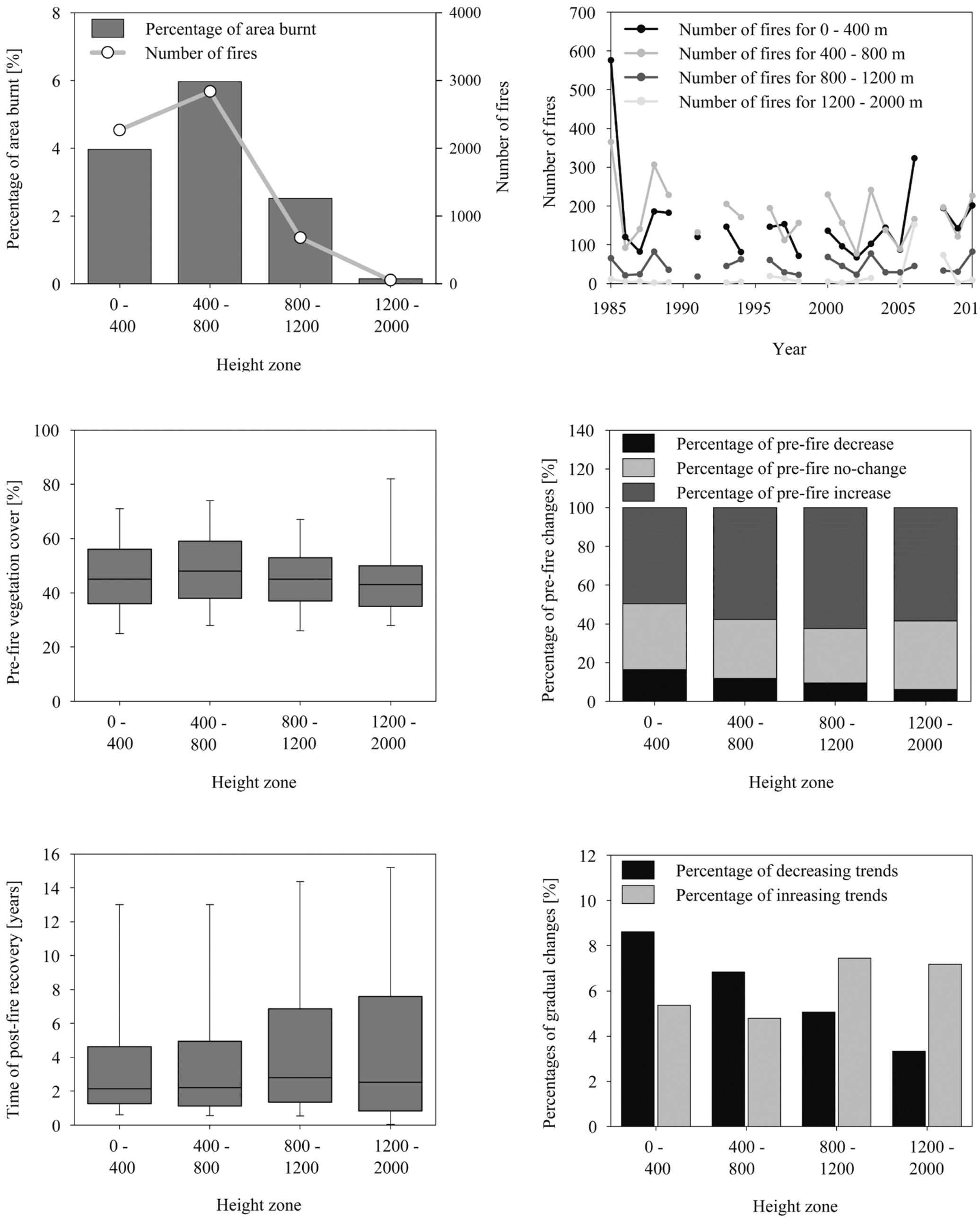

Figure III-7: Dependency of fire and gradual vegetation changes on height: number of fires and percentage of burnt areas, temporal variations of fire frequency between years, fire predisturbance values, direction of fire predisturbance segments, recovery time to reach 50% of pre-fire vegetation cover, percentages of gradual changes.

Burnt areas and gradual vegetation changes differed markedly between the four peripheral units of Crete (Fig. III-8). The number of fires in the prefecture of Chania (2067) was about twice the number of Rethymno, Heraklion (1001 and 849) and higher than in Lasithi (1223), and had the largest percentage of area burnt among all administrative units (5.1% corresponding to 7365 ha). Average pre-fire vegetation cover varied among the administrative units and the two western peripheral units showed higher values than the eastern ones (~ 50% and ~ 40%). The recovery time to reach 50% of the pre-fire vegetation cover generally increased from the western (3.4 years in Chania) to the eastern (4.7 years in Lasithi) peripheral units. All peripheral units were affected by gradual vegetation changes but varied in their proportions of increasing and decreasing trends (e.g., more areas of decreasing vegetation trends in the west). Although, the direction and nature of vegetation changes differed among the peripheral units, the proportion of rangelands affected by changes was overall similar (Chania: 11.4%, Rethymno: 13.3%, Heraklion: 14.9% and Lasithi: 11.8%).

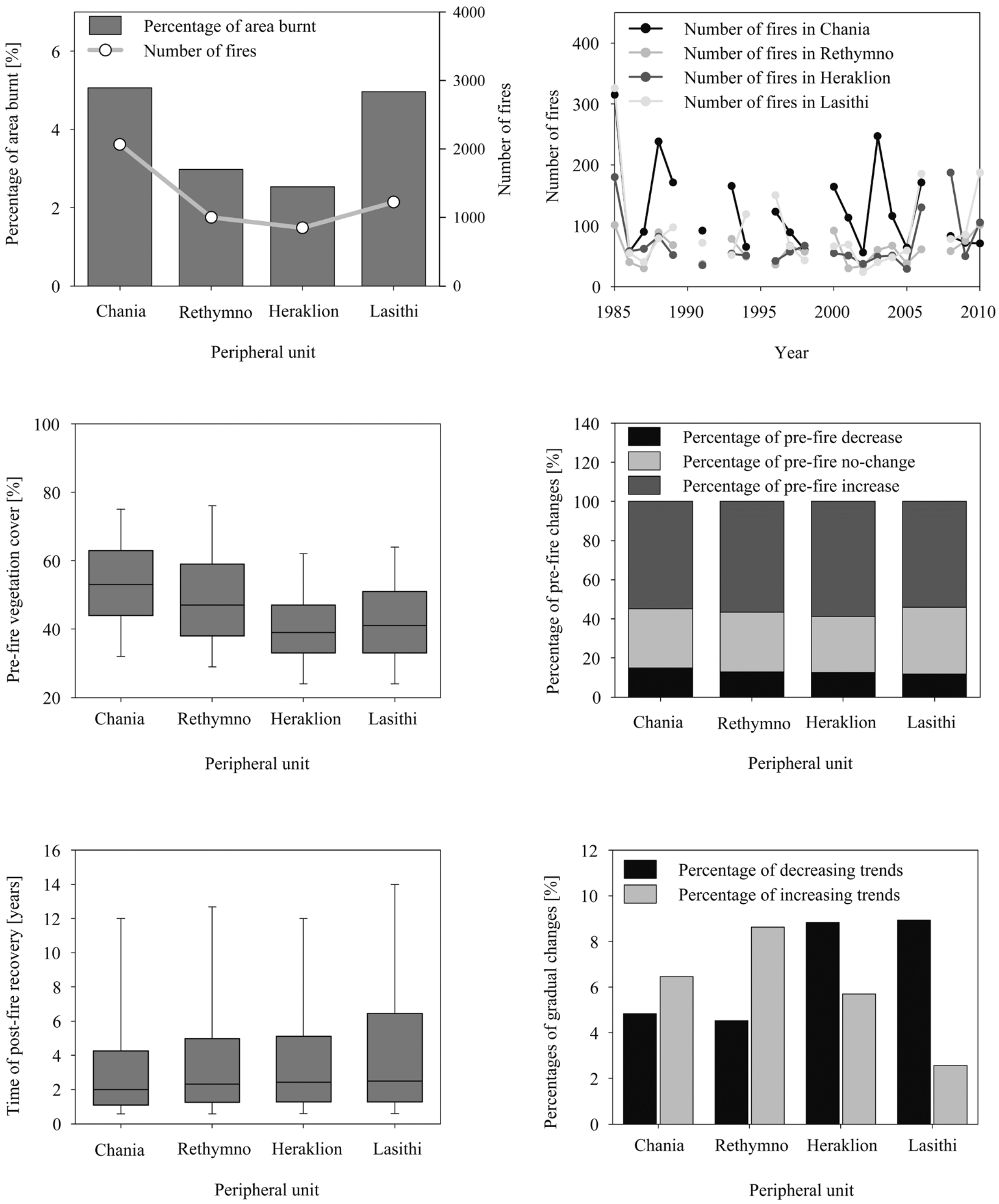

Figure III-8: Dependency of fire and gradual vegetation changes on peripheral units: number of fires and percentage of burnt areas, temporal variations of fire frequency between years, fire predisturbance values, direction of fire predisturbance segments, recovery time to reach 50% of pre7fire vegetation cover, percentages of gradual changes.

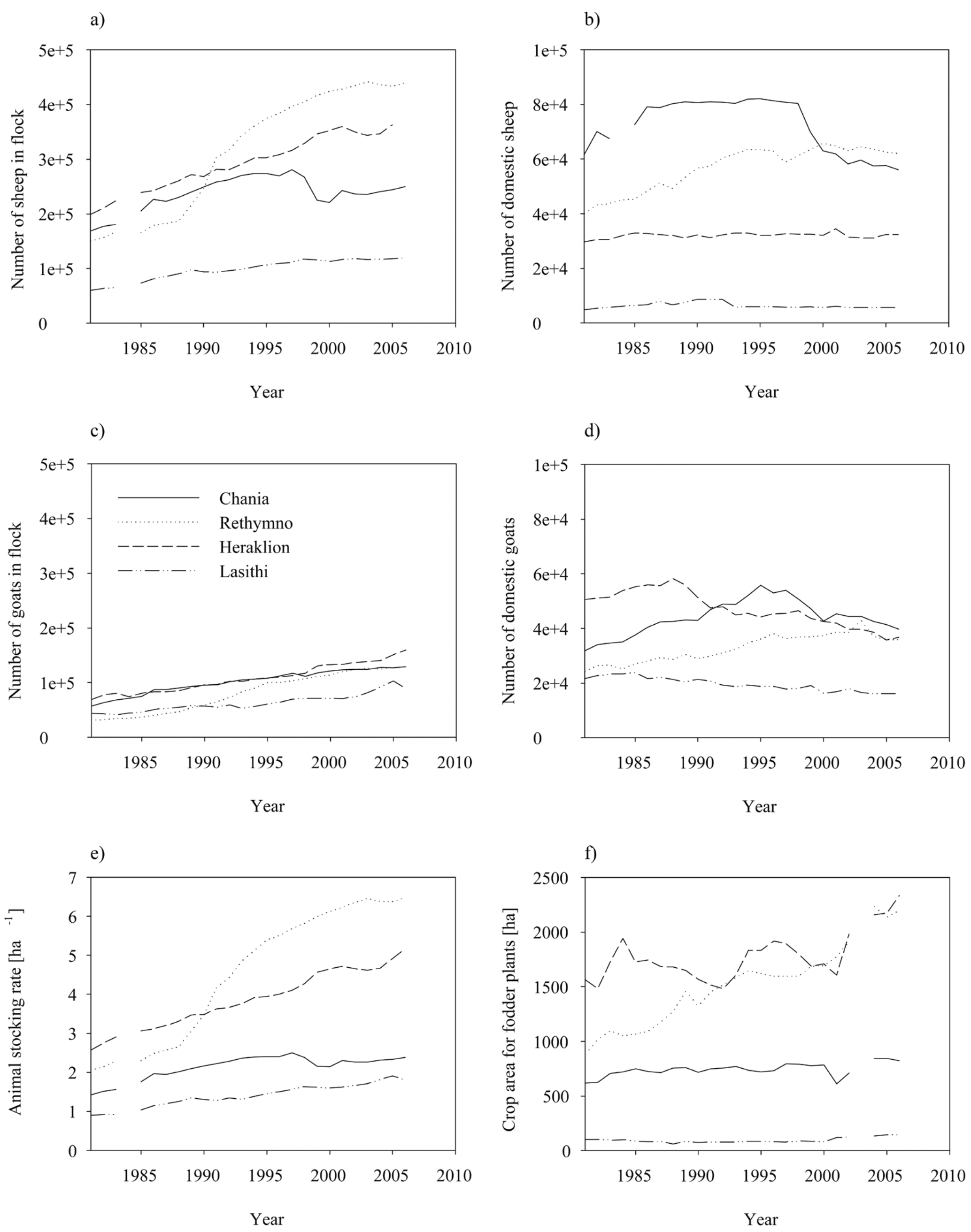

Figure III-9: Changes in the number of animals and kind of livestock breeding for 1981-2006 for the four peripheral units with a) showing changes of sheep in flock, b) domestic sheep c), goats in flock, d) domestic goats, e) representing the stocking rate of sheep and goats in flock and f) showing the area used for planting fodder in Crete.

Total animal numbers of sheep and goats differed substantially among the peripheral units, but generally increased between 1981 and 2006 (Fig. III-9). Livestock was predominantly kept in flocks with sheep by far exceeding the number of goats. While goat numbers increased similarly among all peripheral units, trends in sheep numbers varied. Rethymno showed the highest increase in sheep numbers, especially at the end of the 1980s. In Chania, sheep numbers increased at first, but dropped in 1999 and increased again. Trends in stocking rates for sheep and goats were overall similar to the increases in livestock numbers. Yearly stocking rates varied between ~1 - 2.5 animals per hectare in 1981, but increased to 2 - 6.5 in 2006. Rethymno showed the highest overall increase in livestock rates, followed by Heraklion while the increases in Chania and Lasithi were similar. The area for planting grazing fodder also differed markedly between the peripheral units. The largest area of fodder plants in 1981 was found in Heraklion (1564.9 ha) while Rethymno showed the highest increase in the area used to plant fodder and both peripheral units had similar areas by 2006 (2333.3 and 2196.3 ha for Heraklion and Rethymno respectively).

6 Discussion

Fire frequency and severity have increased over the last decades in many parts of the European part of the Mediterranean Basin. Both changes in climate and land use intensity have been suggested as major drivers of these changes, but their relative importance remains unclear. Regarding land use it is also not clear how the ongoing polarization of land use, with the abandonment of traditional grazing systems on the one hand and intensification on the other relates to vegetation trends and fire regimes. Here, we use a dense time series of Landsat images to reconstruct detailed vegetation cover histories to show that Crete’s rangelands vegetation was substantially affected by fire and grazing alike. Generally, fire regimes appeared mainly driven by changes in grazing pressure, whereas we found only weak evidence for climate effects. Our satellite-based analyses of fire frequency and perimeters also suggests that the number of fires decreased slightly in 1984-2010 with high variations in the annual burnt areas and a decrease in the proportion of large fires, similar to other Mediterranean countries (European Commission 2011), and only partly similar to official Greek fire statistics (http://effis.jrc.ec.europa.eu/fire-history), which suggest a large decrease in fire frequency and extent for our observation period. The trajectory-based analysis of disturbance-like and gradual changes at the same time using dense Landsat time resulted in consistent change results through time and opened the pathway to assess the relationship between grazing induced vegetation changes and fire disturbances in Mediterranean rangelands. Overall, our results suggest that while Crete’s rangelands were influenced by extensive livestock grazing and then fuel limited until recently, fuel accumulation due to the polarization of land use and the abandonment of traditional grazing systems will likely increase the importance of climate as a major driver of fires in the future.

Whereas Crete's climate has become drier and warmer during the last 30 years, we did not observe a strong link between climate variables and fire frequency or areal extent burnt, despite substantial variability in climate variables (e.g., summer rainfall). This weak link suggests that ignitions by lightning are overall scarce and the impact of precipitation on fuel moisture and fuel load is likely superimposed with land use effects controlling fuel availability on Crete. However, although, we did not find a strong link between climate and fires at the scale of the island, we cannot rule out a link between spatial patterns of fires and local climates, especially as topography induces strong orographic effects on rainfall patterns on Crete (Naoum and Tsanis 2003). Although the dependency of large fires on climate is still debated (Moreira et al. 2011), the spatial gradient in precipitation pattern may also explain, in part, the occurrences of large fires in the drier eastern part of Crete in the first half of the observation period.

Grazing appears to be the main driver of vegetation change and fire regimes on Crete. Livestock husbandry has been closely connected with marginal areas in Crete and important for the rural economy. Extensive grazing systems are the most widespread type of livestock husbandry on Crete and principally rely on the indigenous rangeland vegetation. Rangelands are private or managed under communal arrangements with sheep and goats grazing freely within the boundaries of the villages while transhumance includes mountainous pastures nearby. However, livestock numbers increased substantially after the accession to the European Union, when both direct payments per capita and subsidies to support so-called Less Favored Areas became accessible to Crete's farmers (Dubost 1998). As a result, stocking rates increased dramatically for all peripheral units, with changes from 2 up to 6 animals per hectare (Fig. III-8), thereby far exceeding the carrying capacity which is about 1 sheep equivalent per hectare in Mediterranean ecosystems (Papanastasis 1998). Despite these increases in stocking rates, however, we did not find large-scale vegetation decline, but a heterogeneous picture of patches of decreasing and increasing vegetation cover often accompanied by fires (Fig. III-2).

The most plausible explanation for these complex spatial patterns of vegetation trends and fires is the transformation that grazing systems have experienced during the last decades. On the one hand, traditional transhumance declined in importance and is increasingly substituted by the transportation of fodder by car which has been possible by an extension of infrastructure to mountainous areas (Rackham and Moody 1996). At the same time, the overall number of holdings reduced almost by half from 1981 to 2001. As a result, average flock sizes expanded drastically and livestock management intensified on the other hand, thereby increasing the external feed supply with harvested feedstuffs and purchased concentrates although economic unprofitable (Dubost 1998). As a result, the area to plant fodder for grazing increased substantially, from 317 ha to 550 ha in 1981-2006 for Crete (Fig. III-9). Moreover, shepherds also adopt sedentary techniques like the establishment of stalls and watering points. As a result, livestock and grazing pressure concentrates close to the facilities within the village boundaries. Our results clearly reflect spatial pattern of overgrazing and undergrazing at the local scale, often accompanied by fires nearby (Fig. III-2). Moreover, vegetation prior to fires often showed declining and increasing trends, likely the result of intense grazing pressure on the one hand and recovery from a previous fire on the other hand. In addition, slow recovery suggests that livestock graze on these recently burned areas (Fig. III-3).

While we detected widespread fires, the frequency of fires remained relatively stable which is in contrast to the increase in stocking rates and required feedstuff and to what has been the result in other areas in Greece (Xanthopolous 2000). Our results suggest that fuel load is likely the most limiting factor to an increase in fire frequency on Crete in the last decades. On the one hand, the occurrences of fires were strongly related to the initial vegetation cover suggesting that the low vegetated mountainous rangelands were likely fuel limited, mainly the result of shallow soils and long-term impact of livestock grazing. On the other hand, fuel content appeared generally not limited in lowlands and semi-mountainous areas but the concentration of livestock close to stalls is likely accompanied by a similar spatial concentration of fires nearby. Together with slow recovery, their increase is limited. Moreover, it is also possible that fire as a traditional rangeland management tool is becoming less important as grazing systems intensify and the supply of external fodder decreases the dependency on rangeland fodder resources.

Vegetation trends and fire patterns varied also substantially with elevation, likely the result of spatial variation in grazing system transformation and rangeland management. Fire frequency and area burnt for grazing lands showed their maximum between 400 – 800 m followed by a decrease in fire activity with increasing height (Fig. III-7). Likewise, the lowlands were characterized by a higher percentage of decreasing vegetation cover whereas in the highlands more increasing vegetation prevailed. This spatial variation in grazing impacts and fire intensity is likely related to a higher number of intensified systems in lowlands and foothills, whereas the vegetation cover trends we mapped in the mountains highlight a reduction of grazing pressure.

Surprisingly, our results stand in contrast to official fire statistics which indicate a general high number of fires by the beginning of our observation period followed by a strong shift in the fire regime after 1997 and only a few fires afterwards. However, this shift is the direct result of the change in the organization of forest fire management in Greece and subsequent differences in reporting which does not allow a comparison of the official fire statistics of Greece before and after 1997 (Xanthopolous 2000).

Assessments of the link between grazing and fire regimes are largely missing for Mediterranean ecosystems and existing remote sensing based studies focused either on gradual vegetation cover changes, fire occurrences or recovery rates separately. LandTrendr is the a novel semi-automatic trajectory based change detection approach filling this gap in characterizing the entire time series signal of dense Landsat time series, thereby capturing short-term and abrupt changes at the same time (Kennedy et al. 2010). While LandTrendr was developed for forest ecosystems, we here demonstrate that adapting the approach to Mediterranean environments solved the shortcomings of traditional change detection approaches and yielded reliable and consistent results for mapping both gradual vegetation change and fire disturbances. This allowed us to better understanding grazing and fire relationships, and highlight the value of trajectory approaches to reconstruct consistent spatial fire databases from the Landsat archives, which is strongly needed where fire monitoring systems have changed over time, such as on Crete (Grove and Rackham 2001).

While our analyses yielded a reliable vegetation change map with high temporal resolution, some sources of uncertainty remain. First, although phenology of rangeland vegetation did not vary substantially at the level of the island as a whole, vegetation peak dates in the lowlands were up to two months earlier than in the mountains. Less vital vegetation with a higher dry matter content, however, decreases the overall vegetation signal and increases the likelihood that changes are not detected. Second, the maximum number of segments in our LandTrendr implementation was limited to six, which might overlook some fire reoccurrences. Third, we chose a relatively large spatial filter to eliminate mislabeled patches (< 1 ha), however, many fires on Crete occur that are smaller than this threshold, which might explain the differences in our and the official statistics. Last, our validation was mainly based on the interpretation of time series profiles of our Landsat imagery. This approach has been shown to have several advantages over traditional validation approaches that are difficult to implement for trajectory change detection approaches and to produce equally independent assessments of change detection accuracy than using auxiliary data (Cohen et al. 2010; Griffiths et al. 2011). However, difficulties in labeling became apparent where a year was missing in the time series and the area burned was not visually separable from the bedrocks (< 7% of our validation points were affected by such a phenomena).

There is growing concern about the ecological and socio-economic impacts of wildfires in the Mediterranean Basin, particularly in the context of climate change (Pinol et al. 1998; Pausas 2004; Syphard et al. 2009). On Crete, land use appears to be the key driver of fuel availability and connectivity, and the CAP is the most important land use policy in Europe and strongly impacts land use decisions. Current and past policies have resulted in a twofold process of abandonment and fuel accumulation on the one hand, and an intensification and concentration of grazing on the other hand. The recent CAP reform in 2003 decoupled subsidy payments from production and incorporated measures to support traditional land use systems. However, the abandonment of traditional land use activities in the European Mediterranean Basin occurs unabated and at a grand scale (Poyatos et al. 2003; Romero-Calcerrada and Perry 2004; Figueiredo and Pereira 2011). On Crete, our results suggest that fire regimes have been mainly related to land use and we found no evidence for a climate-driven fire regime. However, we detected many signs of an ongoing rural exodus, which is supported by official statistics (e.g., aging rural population, rural population decline (National Statistical Service of Greece 1982-2010)). Together, this strongly suggests an increasing influence of climate on fire regimes on Crete, because fuel accumulation occurs across large areas. The fire regime on Crete will likely shift from a fuel-limited towards a drought driven regime, a process that began in the Western Mediterranean Basin some decades earlier (Pausas and Fernández-Munoz 2010). This is particularly worrisome in light of ongoing climate change, and projections of even hotter and drier summers for the near future (Christensen 2007). The upcoming revision of the CAP in 2013 represents a unique opportunity to counteract current trends in grazing systems (and thus fire regimes) and the potential erosion of the resilience of Mediterranean rangelands.

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Chapter IV:
Analyzing the relative importance of climate, people, and land use in driving fire patterns on Crete (Greece)

Regional Environmental Change (in preparation)

Ruth Sonnenschein, Matthias Baumann and Tobias Kuemmerle

Abstract

Fire is a major disturbance agent in Mediterranean ecosystems. In the Mediterranean Basin, the occurrence and frequency of fires vary strongly across the landscape. Climate factors, the presence of urban dwellers and tourists, and land use potentially all play a role in explaining the spatial patterns of fire frequency, but their relative importance is not fully understood. Moreover, many Mediterranean landscapes are experiencing diverging land use trends, with intensification in some areas and abandonment in others, but the relative influence of these land use trajectories on fire regimes remains largely unexplored. Here, we quantitatively assessed the drivers of fire frequency patterns on the island of Crete (Greece), where strong gradients in climate, population, and land use prevail. Based on a suite of environmental and socio-economic variables, and a combination of best subsets modeling, hierarchical partitioning, and Bayesian Model Averaging (BMA), we tested different hypotheses about the drivers of fire frequency. Our results showed that climate factors and land use, in the form of the abandonment trajectory, were the primary drivers of current fire patterns on Crete. Conversely, the presence of people (urban dwellers and tourists) was not strongly related to fire patters, and we found only weak support for a link between the land use intensification trajectory and fire regimes. The variables selected in our models were consistent across the three approaches we used and across different model dimensionalities. Summer precipitation (positive sign) and the variability in slope (negative sign) both had the highest explanatory power and were always included in our best models. Overall, our results suggest that climate factors and the abandonment of traditional land systems result in more fire-prone landscapes because fire risk increases while fuel accumulation occurs. As both trends are expected to accelerate in the future, fire regimes on Crete will therefore likely shift from a fuel-limited (via grazing) fire regime in the past to a drought-driven fire regime. This will likely increase fire risk for human infrastructure and health, and our results thus underpin the importance of considering agricultural and rural development policies alike in fire mitigation programs.

1 Introduction

Fire is one of the principal drivers of change in the Earth system. Across all biomes, fire is the most widespread and most frequent disturbance event, shaping vegetation patterns from global to landscape scales (Bond et al. 2005; Lavorel et al. 2007; Bowman et al. 2009). It also influences the distribution and structure of vegetation communities, and the occurrence of a wide range of plants adapted to fire or depending on post-fire regeneration (Bond and Keeley 2005; Lavorel et al. 2007; Bowman et al. 2009). Fires also provide an important feedback to the carbon cycle and are a contributor to climate change (Thonicke et al. 2001; Cochrane and Barber 2009), and, perhaps most importantly, fire is a threat to both human settlements and human health (Lampin-Maillet et al. 2010). Understanding the drivers of fire regimes and the spatial patterns of fire occurrence is therefore of great concern.

Climate, human activities, and land use have been held accountable for explaining the occurrence and spatial patterns of fires (Aldersley et al. 2011). However, the relative importance of these drivers often remains unclear. This is worrisome as the magnitude of all these drivers is increasing rapidly, because climate change accelerates, human populations grow, and urbanization and urban sprawl trends continue, and land use change continues to transform natural ecosystems at unprecedented rates (MEA 2005c; Radeloff et al. 2005; Foley et al. 2011). Moreover, the synergies among these drivers remains poorly understood, although these interactions can be strong (Cochrane and Barber 2009; Marlon et al. 2009; Bowman et al. 2011). As global change progresses, there is a tendency towards larger and more frequent fires in many parts of the world (Christensen 2007), highlighting the need to better understand drivers of fire regimes in order to mitigate ecological and socio-economic consequences of changing fire regimes.

Mediterranean ecosystems are especially fire prone due to their distinct climate with hot and dry summers that creates seasonally high fire risk. Although these ecosystems cover only 2% of the Earth's surface, they harbor more than 20% of all vascular plant species (Cowling et al. 1996), and sustain many million livelihoods worldwide (Cox and Underwood 2011). Fires have historically shaped the distribution and structure of vegetation communities in Mediterranean ecosystems, and many plants are specifically adapted to fire, for example via rapid postfire regeneration or morphological and physiological features to withstand fire (Dimitrakopoulos and Panov 2001; Keeley et al. 2011). As many Mediterranean landscapes are increasingly becoming densely populated, fires also more frequently threaten human life and infrastructure, and contribute to a major degradation of ecosystem services (Lampin-Maillet et al. 2010; Shakesby 2011).

The spatial pattern of fires varies strongly among and within Mediterranean landscapes (Marques et al. ; Syphard et al. 2009). Fire patterns are the result of complex interactions between climate, fuel load, and ignition sources. Climate plays a major role because of its direct effects on weather conditions and fuel flammability, and because of direct ignitions via lightning. Moreover, climate indirectly controls fires due to its effects on biomass productivity and thus fuel loads (Pausas 2004). In human-dominated landscapes like the Mediterranean Basin, anthropogenic factors are also important in driving fire regimes (Vazquez et al. 2002). Many ignitions are caused by humans (Syphard et al. 2009), for example, due to recreational activities in rangelands by tourists and weekend travelers (e.g. from urban areas) and the increasing accessibility of previously remote areas due to infrastructure development (Romero-Calcerrada et al. 2008; Catry et al. 2009). Moreover, urbanization in the form of urban sprawl and development of tourism facilities have created larger wildland-urban interfaces, which has been linked to fire patterns in the Mediterranean (Badia et al. 2011).

Fire frequency and patterns are also strongly influenced by land use, as fire is traditionally used as a management tool in many Mediterranean land use systems (Moreira et al. 2011). For example, fires are frequent in rangelands, where shepherds set fires to improve rangelands for livestock husbandry (Catry et al. 2009; Carmo et al. 2011), or on cropland, where farmers use fire to remove crop residues (Bajocco and Ricotta 2008; Catry et al. 2009). Overall, climate factors, an increasing presence of people in rangelands, and land use activities and have been shown to affect the spatial pattern of fires in Mediterranean landscapes, however, the relative importance of these drivers is not fully understood.

In terms of land use, the picture is furthermore more complex because Mediterranean landscapes have been experiencing a diverging trend of agricultural intensification and abandonment over the last decades (Caraveli 2000; Tzanopoulos and Vogiatzakis 2011). Grazing is the main land use activity in Mediterranean rangelands, but livestock husbandry systems have changed drastically. On the one hand, traditional pastoral activities, as well as marginal agricultural land, are abandoned due to their low economic profitability, the high opportunity costs of labor, an over-ageing rural population and a rural population decline due to outmigration (Bernues et al. 2005; Hill et al. 2008). This so-called rural exodus syndrome is particularly prevalent in marginal (e.g., mountainous) landscapes where pastoralism such as transhumance have until recently been widespread (MacDonald et al. 2000). In addition, where rural populations persist there is an increasing tendency away from grazing towards external fodder (Bernues et al. 2005). Together, the abandonment of traditional land use practices results in widespread fuel accumulation in rangeland vegetation communities and an increased fuel connectivity due to the homogenization of the previously highly heterogeneous, mosaic-type landscape patterns (Pausas 2004), overall increasing fire risk (Moreira et al. 2011).

On the other hand, an intensification of livestock farming and croplands has occurred in other regions in the Mediterranean Basin, mainly triggered by subsidies for intensifying production under the European Union's Common Agricultural Policy (CAP, e.g. by headage payments or compensation payments for less-favored areas). As a result, livestock numbers have increased and overgrazing has occurred where stocking rates exceeded rangeland capacity. While this overexploitation of rangeland resources was accompanied by a reduction of fuel availability in some areas, shepherds are also setting fires to increase fodder quality and supply for their growing herds (Xanthopolous 2000; Papanastasis 2004). The intensification process of extensive livestock farming systems has also resulted in a concentration of livestock production and grazing to fewer areas (Bernues et al. 2005), and thus potentially to a concentration of ignitions by shepherds. Yet, analyses of how the divergent land use trajectories of rural exodus and intensification have influenced fire patterns are missing.

Our overarching goal here was to assess the drivers of fire patterns for the island of Crete, Greece. We chose Crete as our study region, because it is characterized by strong gradients in climate, presence of people in the landscape (the island is a prime tourist destination), and land use. In a previous study, we mapped fire frequency for the entire island for the period 2000-2010 (Sonnenschein et al. submitted). Using this map and a community-level database of climate, socio-economic, and environmental variables, we used model selection routines to identify the drives of fire frequency patterns. Specifically, we tested four hypotheses:

4.) The *climate hypothesis* suggests that temperature and precipitation are the primary drivers of fire frequency patterns on Crete, because these parameters are closely linked to fuel availability and flammability.

5.) The *recreational activity hypothesis* suggests that the presence of people in rangelands drives fire frequency patterns, because a large urban-wildlife interface, and a high number of tourists and urban dwellers may trigger a high number of accidental fires.

6.) The *rural exodus hypothesis* suggests that the abandonment of traditional pastoral activities is the primary driver of fire frequency patterns, because fuel availability and connectivity following land abandonment increase fire risk.

7.) The *overexploitation hypothesis* suggests that the intensification of land use is the main driver of fire frequency patterns because intensification leads to a concentration of animals and shepherds, and intentional burning in rangelands.

2 Methods

2.1 Study region

Crete is the largest and most southern island of Greece and the fifth largest island in the Mediterranean Basin, covering an area of 8,265 km². The island is elongated and spans 260 km from east to west and 12 to 60 km from north to south (Fig. IV-1). The terrain is rugged due to several of mountain chains, extending across the island and reaching elevations of up to 2,456 m. These mountain chains are formed of crystalline and platy limestone, while lower mountainous zones are dominated by phyllites, quartzites and younger flysch and marls, sandstone and clays prevail in the plains (Rackham and Moody 1996). Climate is typically Mediterranean, with cool, wet winters and hot, dry summers. Average temperatures vary between 12° C in January and 26° C in July (Hellenic National Meteorological Service 2011). Rainfall mainly concentrates between October and March, but shows great variability across the island due to the strong influence of the relief. As a result, average precipitation ranges from 300 mm in the coastal regions of the southeast to 700 mm in the northwest, and increases to 2000 mm in the mountains (Chartzoulakis et al. 2001).

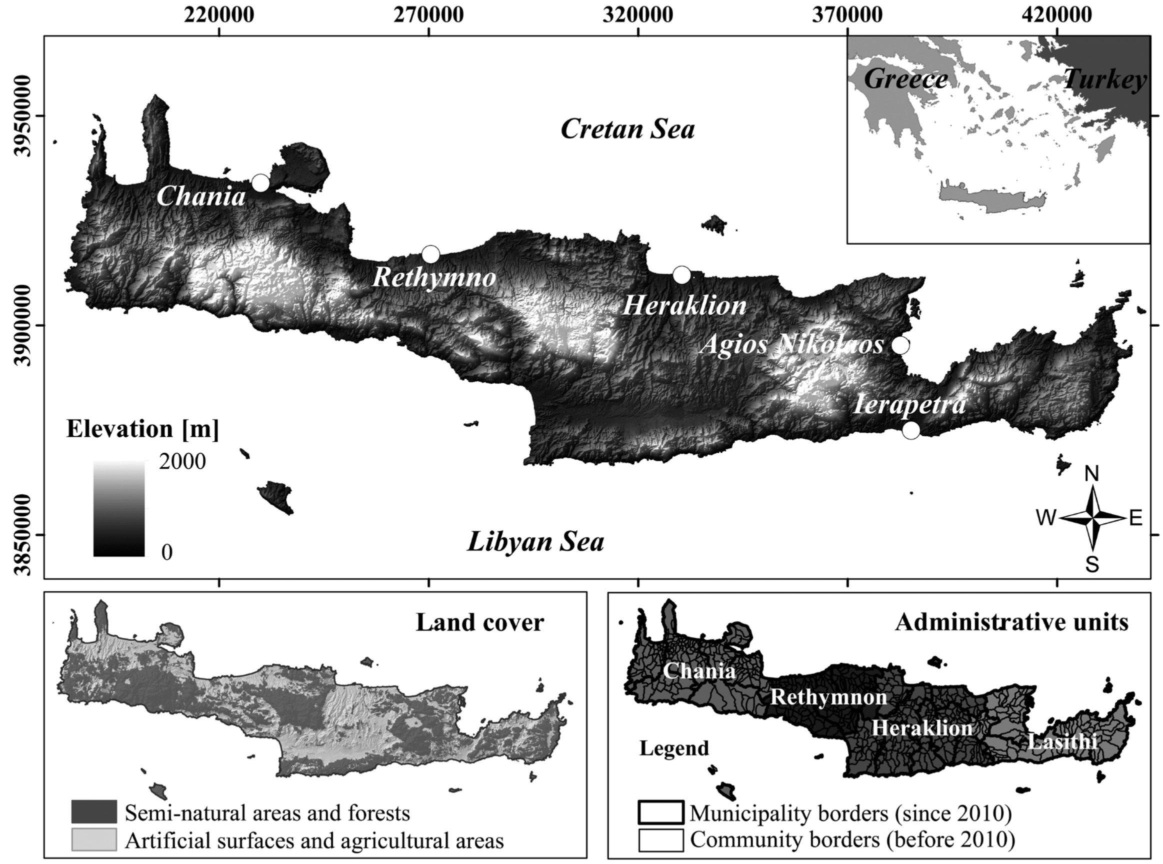

Figure IV-1: The study area of Crete shown as a relief representation and its location within the Mediterranean Sea. The extents of rangelands on Crete are shown as well as the administrative boundaries.

Crete's flora is typical for the eastern Mediterranean region and the result of a long land use history with deforestation and grazing pressure. As a result, potential natural vegetation communities, which are evergreen forests (including species like *Pinus brutia, Cypressus sempervirens* and *Quercus coccifera*), are scarce and reduced to a few stands on the flanks of the mountain ranges. Shrub communities, especially phrygana (i.e., seasonally dimorphic spiny dwarf shrubs such as *Sarcopoterium spinosum, Phlomis fruticosa*) and matoral (i.e., woodland degraded to shrubs) in lowland areas and orophrygana in mountainous areas are the most widespread vegetation types on Crete today. Grasslands are typically found in dolines and flat areas where relatively deep soils prevail (Rackham and Moody 1996).

Greeks administration has been drastically transformed after the accession to the European Union in 1981 and more recently following the administrative reform in 2010 (Kallikrates plan). Crete is now one of the 13 regions (i.e., states) of Greece and is subdivided into four peripheral units (i.e., the former prefectures Chania, Rethymno, Heraklion, Lasithi), which consist of 24 municipalities and replace the 70 municipalities and their lower administrative level of 570 communities. Population mainly concentrates in cities on coastal areas and increased from 502,000 in 1981 to 601,000 in 2001 (NSSG 2009). Likewise, tourism has experienced enormous growth rates, resulting in about 3 million tourists in 2003. Tourism now is the most important economy on Crete, but the tourism impact differs greatly across the island with strong gradients between northern-southern, urban-rural and coastal-hinterland areas, with most tourists concentrated in urban areas at the northern coast (Andriotis 2006).

Agriculture is the main income factor in rural areas, accounting for 13% of the island's Gross Domestic Product and employing about 7% of Crete's labor force. The plains are intensively farmed (mainly olives orchards and vineyards) with an increasing tendency towards establishing irrigation systems and greenhouses since the 1980s. Overall however, livestock breeding in rangelands is the most widespread land use type on Crete and a very important sector of Crete's agriculture. Sheep and goats are mainly farmed for milk production and the majority of grazing systems is extensive with and without transhumance relying on the rangeland vegetation for fodder. While low-lying rangelands are grazed throughout the year, mountainous rangelands are only grazed during summer months. About 20% of the rangelands are state-owned and grazing is a legal right under communal agreements, but land tenure is a complex issue in Crete, frequently causing conflicts among users (Papanastasis 1993). Crete's livestock sector has been greatly transformed after Greece's accession to the European Union and livestock numbers on Crete increased dramatically from 1981 to 2008, with an increase from 713,600 to 1,878,900 sheep and 329,100 to 654,900 goats, while at the same time the number of holdings decreased substantially (NSSG 2009). Shepherds frequently set fire to improve rangeland conditions although it is prohibited during the summer months and burnt areas are excluded from grazing up to 10 years (Papanastasis 1993).

2.2 Fire frequency

To characterize the spatial pattern of fires in Crete's rangelands we relied on a previous analysis which used remote sensing to map fire occurrences (Sonnenschein et al. submitted). In this study, we analyzed both gradual and disturbance-type vegetation cover changes using a near-annual time series of mid-/late- summer Landsat TM/ETM+ images from 1984 to 2010 and a trajectory-based change detection approach (Kennedy et al. 2010; Sonnenschein et al. submitted). Fire occurrences were mapped based on the Normalized Burn Ratio (Key and Benson 2003) and vegetation trends based on Tasseled Cap

Greenness. We generated a per pixel map of fire frequency for the decade 2000-2010. We extracted all fires > 1 ha that occurred in rangelands, using all forests and semi-natural vegetation communities in the CORINE 2000 land cover map (EEA 2005) as our rangeland definition. For each community, we derived average fire frequency (normalized by the rangeland area) across the entire period to account for annual variation. As community boundaries, we selected the administrative borders of 1998. We considered only those 488 communities that contained rangeland areas.

2.3 Explanatory variables of fire regime

We gathered a geodatabase of potential explanatory variables of five groups (Table IV-1): (1) climate, (2) anthropogenic influence (other than land use), (3) rangeland management, (4) rangeland conditions, and (5) topography. We used the year 2000 (i.e., the beginning of our observation period) as our base year. Some of our indicators where available for multiple points in time. We calculated changes in these indicators for the period *preceding* our fire frequency mapping in such cases. We used the preceding period to capture legacy effects (e.g., more fires because of prior fuel accumulation) and because our goal was to explain spatial patterns in fire frequency patterns, not changes in fire frequency over time (temporal trends in fire frequency are negligible for the time period studied).

Climate

Our first group included precipitation and temperature variables. Climate data were available as daily, gridded observations from the re-analysis of meteorological observations from September 1957 to August 2002 (ERA-40 data set) produced by the European Centre for Medium-Range Weather Forecasts (ECMWF) (Uppala et al. 2005), downscaled to 1 km (Holt 2005). We calculated (1) average yearly and summer (June, July, August) mean air temperature, (2) annual and summer rainfall, and (3) trends for the period 1981-1999 for each community.

Table IV-1: Potential explanatory variables used in our study. Variables that were collinear and excluded from the following analyses are marked with *

Indicator	Variable	Time
Climate		
Precipitation	Mean summer precipitation [MEAN_PS]	1981-1999
	Mean summer precipitation trend [TREND_PS]	1981-1999
	Mean annual precipitation [MEAN_PY]	1981-1999*
	Mean annual precipitation trend [TREND_PY]	1981-1999*
Temperature	Mean summer temperature [MEAN_TS]	1981-1999
	Mean summer temperature trend [TREND_TS]	1981-1999
	Mean annual temperature [MEAN_TY]	1981-1999*
	Mean annual temperature trend [TREND_TY]	1981-1999*
Anthropogenic influence		
Rural population	Population density [POP_DENSITY]	1999, 1999-1991, 1999-1981*
Households	Household density [HH_DENSITY]	1999, 1999-1991*
Remoteness	Distance to towns [DIST_CITIES_U10K]	
	Distance to cities [DIST_CITIES_O10K]	
	Distance to coastline [DIST_COAST]	
Accessibility	Road network density for 1st, 2nd and 3rd level streets [ROAD_DENSITY]	2011
Rangeland management		
Livestock stocking rates	Stocking rates for sheep [SHEEP_SR]	1999, 1999-1991*, 1999-1981*
	Stocking rates for goats [GOATS_SR]	1999, 1999-1991*, 1999-1981
Holding density	Holdings per ha [FARM_DENSITY]	1999, 1999-1991
Orientation of holding	% livestock holdings [PERC_L_FARM]	1999, 1999-1991*
Livestock capital of holding	Livestock per holding [LST_PER_FARM]	1999, 1999-1991
		1999, 1999-1991*
Total labor force	Livestock per labor force [LST_PER_LABOR]	1999, 1999-1991
Labor force composition	% family labor force [PERC_F_LABOR]	1999
Land tenure	% state-owned rangelands [PERC_MUN_RL]	2000
Fodder supply	% of rangelands [PERC_RL]	
Rangeland conditions		
Soil type in rangelands	% of Leptosol [PERC_I]	2004
	% of Luvisol [PERC_L]	
	% of Regosol [PERC_R]	
Vegetation cover	Mean vegetation cover	1984-1999
Grazing impact	[MEAN_VEGETATION]	1984-1999
	% undergrazed rangelands [PERC_GROWTH]	1984-1999
	% overgrazed rangelands [PERC_DEGRADATION]	
Topography		
Elevation	Median elevation [MEDIAN_ELEVATION]	2000
	Standard deviation of elevation	2000
	[STD_ELEVATION]	2000
Slope	Median slope [MEDIAN_SLOPE]	2000
	Standard deviation of slope [STD_SLOPE]	2000
Aspect	Median northernness [MEDIAN_NORTH]	

Anthropogenic influence

Our second group of variables proxied the presence of people and accidental ignitaions related to their activities. We derived population density and household density for each community from the population censuses that were carried out in 1981 (National Statistical Service of Greece 1981b), 1991 (National Statistical Service of Greece 1991) and 2001 (National Statistical Service of Greece 2001). We also calculated two measures for the remoteness of communities by calculating distances from the community center (i.e., centroid) to the nearest towns and to the five major urban centers on Crete based on cartographic maps. The latter is in an important measure for the prevalence of tourism as four of the major cities have harbors and two have international airports. Additionally, we calculated the closest distance to the coastline of each community’s center, because most tourism is concentrated on the coast. To approximate accessibility of rangelands within a community, we also derived road density within each community by dividing the length of primary, secondary, and tertiary streets by the community area (http://www.openstreetmap.org).

Rangeland management

Our third group involved variables related to rangeland management, which we assumed to directly affect fire frequency, because shepherds set fires and because the intensity of grazing as well as the temporal and spatial allocation of grazing controls the available of fuel. We took most rangeland management variables from the Basic Survey for the Structure of Agricultural Holdings 1999/2000 (National Statistical Service of Greece 2004) and calculated changes to the Agricultural-Livestock Census carried out in 1991 (National Statistical Service of Greece 1998) and in 1981 (National Statistical Service of Greece 1981a). We calculated all change variables considering all changes in community codes and administrative boundaries (http://dlib.statistics.gr).

To approximate grazing pressure by livestock type, we extracted the number of sheep and goats per rangeland area and calculated stocking rates. To capture the management intensity of livestock farming systems, we gathered information on the primary orientation of holdings, divided into livestock breeding and mixed farming (agricultural and livestock breeding) holdings and calculated the density of total holdings per available rangeland area and the percentage of livestock holdings. Moreover, we assumed that mixed holdings substitute rangeland fodder resources by feedstuff to a greater extent than livestock holdings which have to purchase external fodder. We also assumed that farm size (i.e., overall livestock numbers) affects rangeland management practices. Therefore, we calculated the average number of goats and sheep for the livestock and mixed holding. We also derived two variables to capture labor intensity in the livestock husbandry sector: the ratio of livestock numbers to the overall number of workers in livestock husbandry per community and the ratio of family members employed at the holding to total work force.

We obtained information on land tenure and calculated the percentage of state-owned rangelands per community. Where rangelands are used as common pool resources and no regulation exists regarding the spatial and temporal allocation of grazing, the result is often both overgrazing and undergrazing (Papanastasis 2009), which both may alter fire patterns. To derive a measure of rangeland fodder resources and external fodder supply, we calculated the percentage of rangeland areas within the boundaries of each community, which is inversely related to the area of arable land where fodder plants for grazing are planted.

Rangeland conditions

Our fourth group of variables approximated the condition of rangeland vegetation, which determines both fuel availability and forage productivity. To measure fuel availability, we derived average vegetation cover estimates for the period 1984-1999 based on remote-sensing based trends in vegetation from our previous study (Sonnenschein et al. submitted) and calculated mean cover values for each community. We also included gradual changes in vegetation cover between 1984 and 1999, which are assumed to represent changing grazing pressure (Hostert et al. 2003a; Sonnenschein et al. 2011), by extracting extents of vegetation cover decline and increase or each community for the period. We only considered areas that had changed by more than 5% cover (decline or increase). Because soil types can influence vegetation growth and fuel moisture content, we also derived the share of the three most prevalent soil types on Crete (leptosol, luvisol and regosol) for each community using the 1:100,000 European Soil map (European Commission 2004).

Topography

Our final group of variables related to topography, which strongly influences local climate conditions and vegetation patterns. Mountainous rangelands serve only as a fodder source in summer after snow cover has melted and elevation thus influences the temporal and spatial distribution of grazing in transhumance systems. Steep slopes inhibit shepherds and their flocks to access grazing areas. Aspect determines the effect of solar heating which has a strong effect on plant species distribution with south facing slopes dominated by xeric exposures and north facing slopes by mesic exposures. To derive topography-related variables, we obtained the SRTM data (Shuttle Radar Topography Mission, http://www2.jpl.nasa.gov/srtm) and derived median and standard deviation elevation and slope for all rangelands areas within each community. We also calculated a continuous variable measuring northerness as:

$$N = \cos - (\frac{A * \pi}{180}) \qquad (1)$$

where N is northerness, ranging from 0 (south) to 1 (north) and A is aspect in degrees. We summarized the northerness index at the community level by calculating the median for all rangeland pixels.

In total, we derived 46 candidate explanatory variables: 8 for climate, 9 for anthropogenic influence, 18 for rangeland management, 6 for rangeland conditions, and 5 for topography (Table IV-1).

2.4 Modeling approaches

To quantify the relative importance of our explanatory variables in explaining fire frequency patterns on Crete, we used a combination of best subsets regressions, hierarchical partitioning, and Bayesian Model Averaging. All analyses were carried out in the statistical software package R (v2.14, http://www.r-project.org).

First, we assessed correlations between our explanatory variables by calculating Pearson's R^2 for each variable pair and excluded one variable from collinear variables pairs for $R^2 > 0.65$. Annual and summer temperatures and precipitation were highly collinear and we retained only summer temperature and precipitation variables as we assumed these variables to be more directly related to fire patterns. , variables approximating rangeland management for different points in time were often correlated and we kept the more recent one (e.g. stocking rates for sheep and goats for 1999, Table IV-1) in such cases. Finally, we also found the number of livestock per holding and the number of livestock per total labor force to be collinear. Both variables approximate the intensity of livestock farming systems and we kept the first variable due to its higher explanatory power to the dependent variable. In total, we dropped 11 variables from our analyses due to collinearity (Table IV-1).

Our statistical analysis involved three levels. On the first level we evaluated the importance of our explanatory variables across the best models, by counting how often a variable appears in the best models of each model complexity (model complexity refers here to the number of variables in the models). To find the best models, we used best subset regression. Best subset regression performs an exhaustive search among all regression models and ranks all regression models based on a measure of fit (in our case the Bayesian Information Criterion (BIC), Schwarz 1978; Miller 2002). We limited our model complexity to 12 variables and derived the best 25 models for each model complexity. We then summarized how often an explanatory variable was selected per model dimensionality.

In the second analysis level, we evaluated the relative importance of each explanatory variable for each model complexity. We used Hierarchical partitioning (HP) to quantify the contribution of a variable to the fit of each regression model from the best subsets routine by comparing the fit of a model with and without the explanatory variable (Chevan and Sutherland 1991). HP calculates these differences for each model complexity and averages the single results to derive the independent contribution of a variable within a model (Mac Nally 2002). This yielded a measure of variable importance for each variable and model dimensionality.

In the third analysis level, we evaluated the overall importance of our explanatory variables across all model complexities. We used Bayesian Model Averaging (BMA) because of its strength to consider multiple models (Hoeting et al. 1999). The basic idea of BMA is that two hypothetical models can have a similar fit while containing completely different explanatory variables and different standard errors. In other words, instead of considering only one model, BMA aggregates multiple similar performing models, yielding overall better predictive abilities (Raftery et al. 1997). We used the model selection, created by the best subset regression analysis, and calculated for each model the posterior probabilities of the models assuming uniform priors for the coefficients. We then used the Occam's razor (OZ) to select the models that were best supported by the data. We used the default OZ (i.e., 20) the BMA package in R (Raftery et al. 2006). Using the same package, we also calculated the coefficient posterior probabilities, which indicate whether a coefficient is different from zero, as well as the posterior mean and standard error for each coefficient (Hoeting et al. 1999). From this process we obtained coefficients for our explanatory variables that allowed us to interpret the influence of each variable across the models.

3 Results

The type of variable selected in the 25 best models varied across model dimensionalities in the best subsets analyses (Fig. IV-2). From the 35 explanatory variables in our predictor database, 15 were always selected, and six additional variables were selected in at least three model dimensions. Several variables were rarely selected, for example *percentage of livestock holdings in 1999*, *number of livestock per holdings in 1999*, *percentage of municipality rangelands, percentage of rangelands*, *percentage of regosol*, *median elevation*, *median north* and *road density*. Some variables were never selected, for example variables concerning rangeland management (*change in livestock per holding* and *change in livestock per labor force between 1999 and 1991*) and anthropogenic influence (i.e., *household density in 2001*, *distance to towns* and *distance to the coastline*) (Fig. IV-2).

Trend in summer precipitation was consistently selected across all model dimensions and included in the majority of the best subset models with low dimensionality (i.e. 2-4 variables, 22/25, 24/25, 22/25). The number of selections of this variable decreased, however, with increasing model dimensionality (i.e. 10-12 variables 12/25, 6/25, 14/25). Low dimensional models tended to select environmental variables (i.e., *trend in summer precipitation*, *standard deviation of slope*, *population density in 2001*, *percentage of vegetation decline*, *percentage of leptosol*), while rangeland management variables (i.e., *stocking rate of goats* and *farm density in 1999*) were only selected in higher dimensional models. Overall, the selection of variables was relatively consistent among model dimensions, and those variables that were once selected were retained across dimensions while other explanatory variables were successively added (Fig. IV-2 and Fig. IV-3).

As can be expected when assessing complex human-environment systems, the explanatory power of our regression models was modest, ranging from a mean adjusted R^2 of 0.043 for 2-dimensionals models to a mean adjusted R^2 of 0.108 for 12-dimensional models. Model fits were very similar across the 25 best models within each model dimension and adjusted R^2 differed less than 0.014 (differences in BIC = 8.479) for bivariate regression models and showed decreasing variability with increasing model dimension (differences in adjusted R^2 = 0.005 and BIC = 2.982 for 12-dimensional models) (Fig. IV-4).

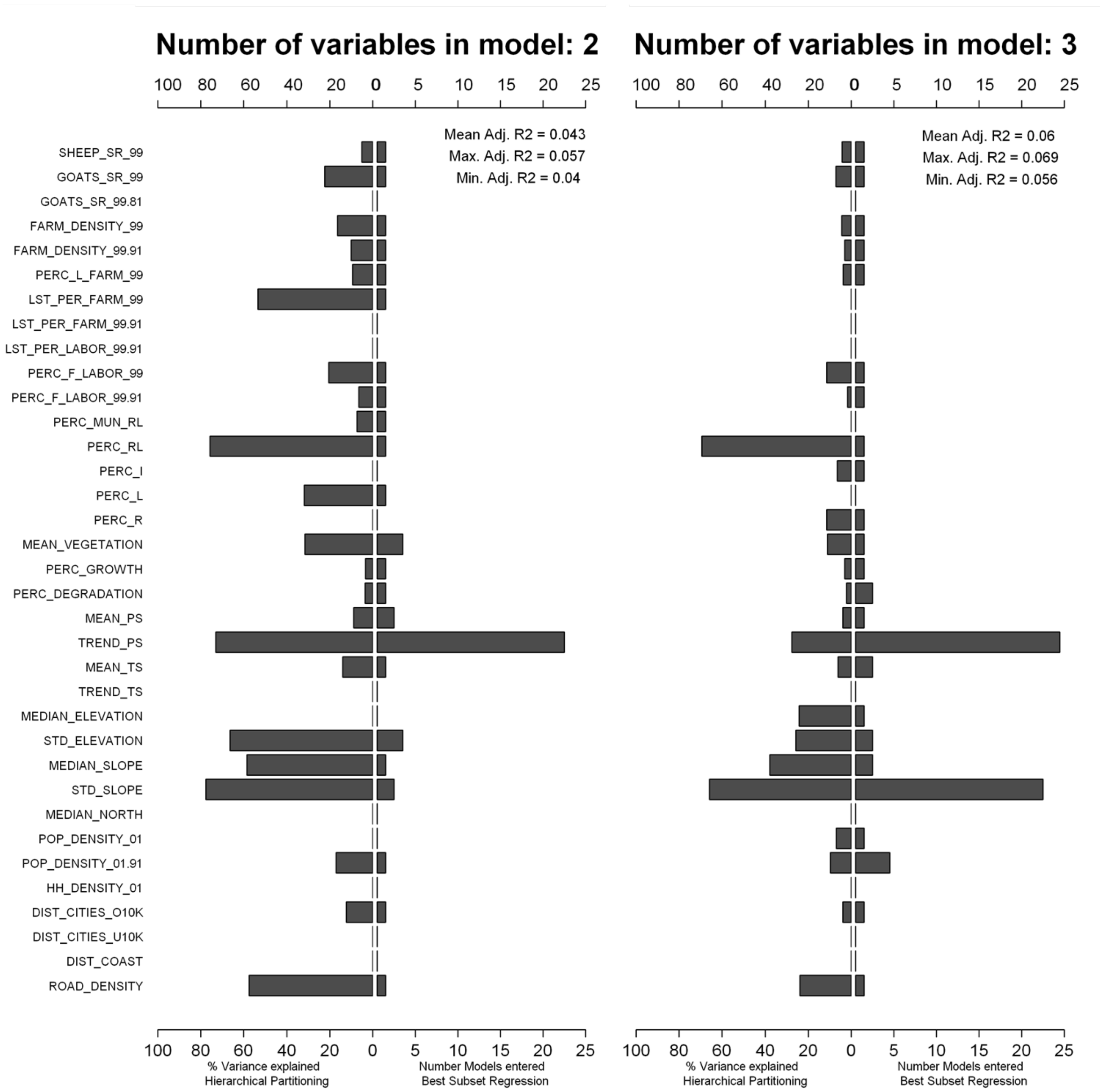

Figure IV-2: Summary results of the best subset regressions and hierarchical partitioning for 2- and 3-dimensional models: Average independent contribution of the predictor variables, times they entered into the best regression models, and goodness of fit measures.

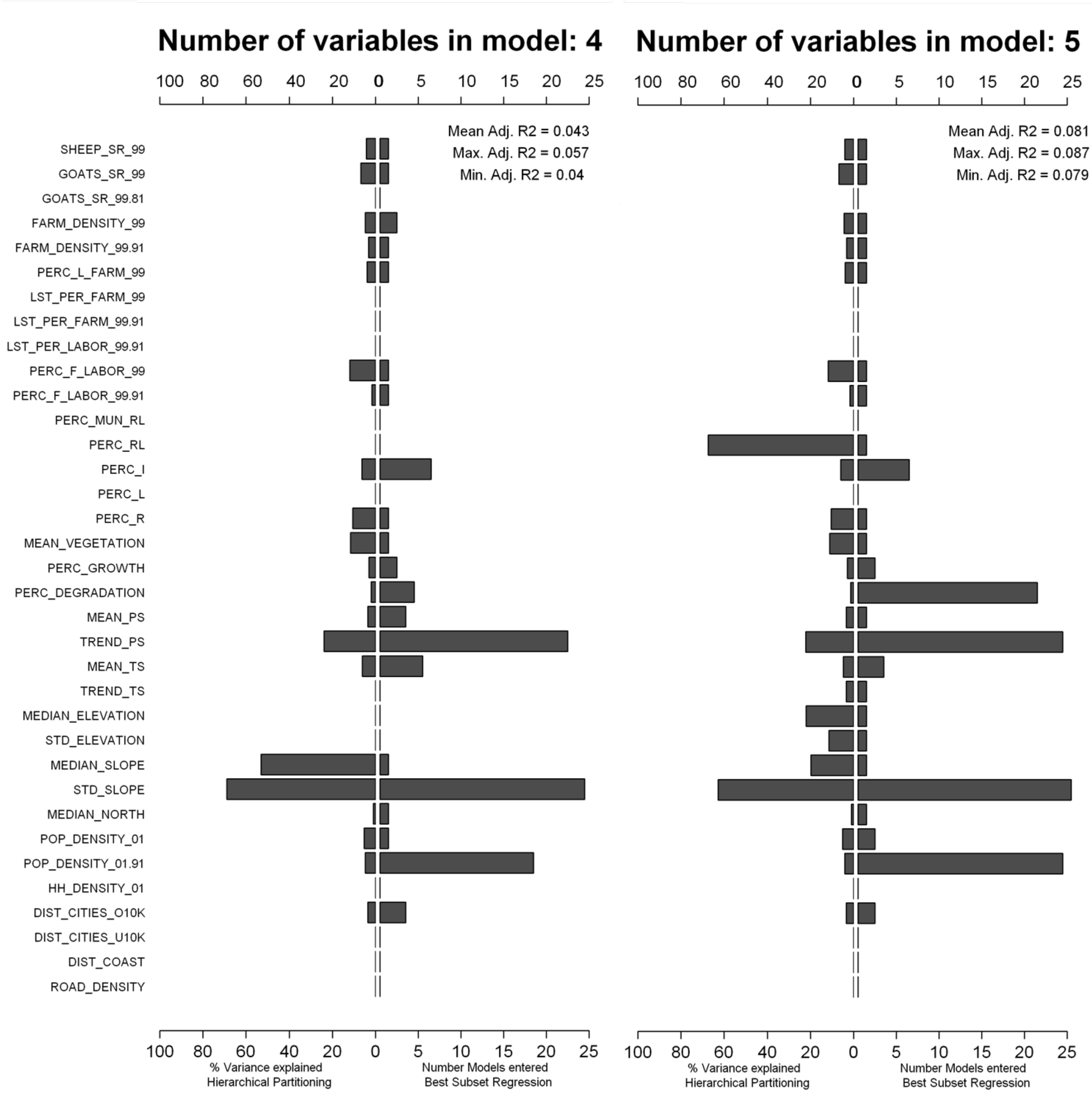

Figure IV-3: Summary results of the best subset regressions and hierarchical partitioning for 4- and 5-dimensional models. Average independent contribution of the predictor variables, times they entered into the best regression models, and goodness of fit measures.

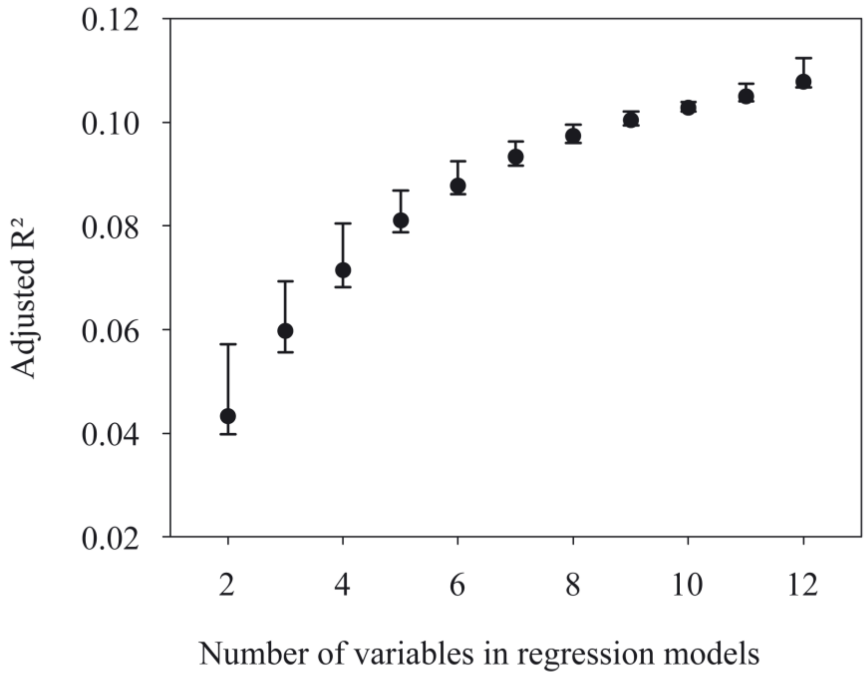

Figure IV-4: Goodness of fit for all model dimensions.

The hierarchical partitioning showed that the relative explanatory power of variables varied substantially. Several variables contributed at least with 50% to the total explained variance in fire frequencies among communities in bivariate models (*trend in summer precipitation* (73.0%), *road density* (57.4%), *percentage of rangelands* (75.7%), *standard deviation of elevation* (66.4%), *median slope* (58.6%), and *standard deviation of slope* (77.6%)). With increasing model dimension, the number of variables with high relative contributions to the explanatory power decreased and only one or two variables contributed relatively high to the overall explained variance in subsequent multivariate models while their overall relative contribution decreased. (Fig. IV-2, Fig. IV-3)

The Bayesian Model Averaging confirmed the results from the best subset regression modeling. Overall, 72 models were selected in the BMA (Table IV-2). *Standard deviation of slope* and *trend in summer precipitation* showed overall highest posterior probabilities that their coefficients were not zero among all variables (95.5% and 94.5%, respectively) and we found negative coefficients for *standard deviation of slope* and positive coefficients for *trend in summer precipitatio*n. Likewise, we found relatively high posterior probabilities for *population density change between 2001 and 1991* (70.5%) with negative coefficients, followed by *percentage of vegetation decline* (43.5%) and *percentage of leptosol* (21.5%) which both had positive coefficients. The majority of variables had posterior probabilities of less than 5% or exactly zero. Correspondingly, mean and standard deviation values of *standard deviation of slope* coefficient's posterior distribution were -0.125 and 0.0496 while for *trend in summer precipitation* mean value was 0.0111 and standard deviation 0.00387 while those of other variables were smaller or zero.

Table IV-2: Results of the models selected by Bayesian Model Averaging. p!=0 is the posterior probabilities that the coefficients are not zero, EV and SD are the mean and standard deviation values of each variable's coefficient's posterior distribution. For each of the five best models the coefficients are shown as well as the model parameters.

	p!=0	*EV*	*SD*	*Model 1*	*Model 2*	*Model 3*	*Model 4*	*Model 5*
Intercept	100	-2.62E+00	1.17E+01	0.832231	1.074336	1.062811	0.923665	1.167516
MEAN_PS	5.3	1.58E-04	6.94E-04					
TREND_PS	94.7	1.11E-02	3.87E-03	0.012495	0.011585	0.010167	0.012047	0.011134
MEAN_TS	11.8	1.46E-03	4.77E-03					
TREND_TS	0.6	1.13E-05	2.87E-04					
POP_DENSITY_01	5.4	4.27E-05	2.08E-04					
POP_DENSITY_01.91	70.5	-5.10E-03	4.15E-03	-0.0074	-0.00643		-0.00703	-0.00606
HH_DENSITY_01	0	0.00E+00	0.00E+00					
DIST_CITIES_U10K	4.4	0.00E+00	0.00E+00					
DIST_CITIES_O10K	0	4.73E-07	2.71E-06					
DIST_COAST	0	0.00E+00	0.00E+00					
ROAD_DENSITY	0.9	-2.31E-06	2.90E-05					
SHEEP_SR_99	3.5	-1.89E-07	1.20E-06					
GOATS_SR_99	4.6	-8.14E-07	4.43E-06					
GOATS_SR_99.81	0	0.00E+00	0.00E+00					
FARM_DENSITY_99	4.9	-1.20E-05	6.19E-05					
FARM_DENSITY_99.91	1.5	4.18E-06	4.85E-05					
PERC_L_FARM_99	1.4	-2.25E-04	2.65E-03					
LST_PER_FARM_99	0	0.00E+00	0.00E+00					
LST_PER_FARM_99.91	0	0.00E+00	0.00E+00					
LST_PER_LABOR_99	0	0.00E+00	0.00E+00					
PERC_F_LABOR_99	4.4	-2.26E-04	1.27E-03					
PERC_F_LABOR_99.91	3.3	-1.19E-04	7.86E-04					
PERC_MUN_RL	0	0.00E+00	0.00E+00					
PERC_RL	0.5	9.38E-06	2.95E-04					
PERC_I	21.5	9.03E-04	1.95E-03				0.00387	0.003743
PERC_L	0	0.00E+00	0.00E+00					
PERC_R	3	-9.41E-05	6.63E-04					
MEAN_VEGETATION	1.6	1.34E-04	1.66E-03					
PERC_GROWTH	4.8	-4.99E-04	2.64E-03					
PERC_DEGRADATION	43.5	1.48E-02	1.92E-02	0.034884			0.035588	
MEDIAN_ELEVATION	0.5	7.53E-07	2.72E-05					
STD_ELEVATION	1.8	-2.00E-05	1.87E-04					
MEDIAN_SLOPE	1.1	-2.47E-04	3.32E-03					
STD_SLOPE	95.5	-1.25E-01	4.96E-02	-0.12286	-0.11787	-0.10862	-0.16434	-0.15791
MEDIAN_NORTH	0.6	6.74E-06	1.44E-04					
Number of variables				4	3	2	5	4
R^2				0.088	0.075	0.061	0.097	0.083
BIC				-18.4797	-17.9768	-17.0869	-16.5857	-15.7535
Posterior probability				0.105	0.082	0.052	0.041	0.027

The five best such weighted models showed a cumulative posterior probability of 0.3067 and consisted of two to five dimensional models. The overall best model had a posterior probability of 0.105 and the lowest BIC (-18.48). This model included four predictors: *standard deviation of slope*, *trend in summer precipitation*, *change in population density between 2001 and 1991* and *percentage of vegetation decline*. Posterior probabilities and goodness of fit decreased successively from 0.082 to 0.027 (BIC: -17.98 to -15.75) for the second- to fifth-best model. The variables *standard deviation of slope* and *trend in summer precipitation* were also selected in these models while *change in population density* *between 2001 and 1999* was excluded in the bivariate model that was ranked as the third best model. The fourth- and fifth- best models additionally included *percentage of leptosol* in combination with *percentage of vegetation decline* (fourth-best model) and without this variable (fifth-best model).

4 Discussion

Climate, people, and land use are primary drivers of fire patterns, but the relative importance of these drivers is not fully understood. The Mediterranean Basin is a fire-prone and human-dominated region, where climate, human presence in wildlands, and land use are all changing markedly, and increasingly influence ignitions and fuel availability. Understanding the linkages between climate, people and land use in driving fire patterns is therefore particularly important for this region. Moreover, resolving the role of land use in determining fire patterns is not easy as the diverging land use trends of abandonment and intensification co-occur. Here, we assessed four hypotheses regarding the relative importance of climate factors, recreational activities, land use abandonment, and land use intensification for determining fire frequency patterns on Crete. Our results suggest that climate factors and rural exodus are the principal drivers of fire frequency patterns on Crete, whereas the mere the presence of urban dwellers and tourists, and land use intensification did not explain fire frequency patterns well. This suggests that the at present fuel-limited fire regime will likely change into a drought-driven fire regime, which increases fire risk and thus risk for harm to human infrastructure, health, and a degradation of ecosystem services.

Our statistical models provided support for the climate hypothesis, suggesting a marked link between climate parameters and fire frequency patterns. Especially the variable *trend in summer precipitation* was positively correlated to fire frequency and was included in the majority of the models (95%) selected by our Bayesian Model Averaging algorithm. This can be explained by the documented dual effect that precipitation has on flammable fuel content (Pausas 2004). First, high precipitation results in high biomass productivity and as such in high fuel content during the dry season. Second, increases in summer precipitation in the previous decades likely resulted in a more homogeneous pattern of fuel as more dense vegetation stands develop (e.g., relaxing water stress). These views are further supported by the relatively high, and positive influence of mean summer temperature suggesting fires were especially frequent were the legacy effects of increasing precipitation (i.e., more fuel) and higher summer temperatures (i.e., low levels of fuel moisture leading to higher flammability) co-occurred. In general, these results are consistent with other studies in the Mediterranean Basin, showing that higher precipitation is positively correlated with both fire frequency and the burnt area although these studies focused on concurrent variations in climate and fire frequencies (Pinol et al. 1998; Pausas 2004). Taking a closer look at the role of climate variables in explaining fire frequency patterns in our study, however, also reveals that climate factors alone do not fully explain the fire patterns we observed. For example, average mean precipitation was only weakly related to fire frequencies. This suggests that other factors, related to human activities, also play a substantial role in determining fire patterns on Crete.

Among human activities, we found only weak support for the recreational activity hypothesis (i.e., high fire frequencies mainly due to accidental ignitions by urban dwellers and tourists). Although population density change was important (e.g., included in 70.5% of all the models in the best subsets routine and included in four out of the five best BMA models), the influence of this variable was opposite to our expectations. *Changes in population density between 2001 and 1991* was negatively correlated to fire frequency patterns, in other words fires were more frequent where population density declined. Accessibility, a proxy for the presence of urban dwellers and tourists in rural areas and thus accidental ignitions, was not strongly related to fire frequency and the variable *road density* was included rarely in lower dimensional models. These results disagree with previous studies in the Mediterranean Basin showing a strong influence of population density and fire occurrences (Romero-Calcerrada et al. 2008; Catry et al. 2009). Likewise, explanatory variables approximating tourist influence also did not show strong linkages to the fire frequency patterns on Crete, although tourist presence is linked to ignitions elsewhere in the Mediterranean Basin (Badia et al. 2011).

To the contrary, our results strongly support views that the abandonment of traditional land use systems is a major driver of fire patterns on Crete. Especially, the negative correlation between population decline and fire patterns suggest a linkage between fire regimes and rural exodus developments on Crete. Despite considerable support for marginal areas (e.g., via the Common Agricultural Policy), Crete, like many areas in the Mediterranean Basin, is facing an emigration of young people from rural areas and a lack of successors in livestock farming due to the hard working conditions and comparatively low income opportunities, leading to an abandonment of traditional transhumance systems (Dubost 1998). Areas where rural populations are declining are thus characterized by fuel accumulation, which as confirmed by the remote sensing analyses where we found strongly increasing vegetation trends for many such communities (Sonnenschein et al., submitted). Our rural exodus hypothesis (i.e., higher fire frequencies in areas where traditional grazing systems are abandoned) was also supported by the negative sign of the topography variables (e.g., mean and standard deviation of slope). The relative influence of these variables were similarly strong as in the case of some of the climate variables, suggesting fires were more frequent in steeper and more rugged areas. Such areas are generally difficult to access, are only used in traditional land use systems, and thus likely the first areas to be abandoned.

Despite substantial support for our rural exodus hypothesis, however, it was surprising that our rangeland management variables explained fire frequencies only relatively weakly. One reason for this might be that the influence of management is indirectly captured by other variables, most likely by topography (i.e. slope and elevation). Moreover, our socio-economic data was only available aggregated at the community level, which may have been too coarse to establish a strong link between land management variables and the fire patters we observed, and which ignores spatial heterogeneity within communities (e.g. livestock stocking rates can vary within the rangelands within one community, Röder et al. (2007)).

Although high fire frequency appeared relatively strongly related to the rural exodus syndrome, we also found some evidence for linkages between high fire frequencies and the overexploitation hypothesis (i.e., high fire frequency due to a concentration of grazing, and thus fires set by shepherds). The percentage of degraded rangelands (measured by the level and trend in vegetation cover, Hostert et al. 2003, Sonnenschein et al, submitted) was selected in our BMA models and had a positive coefficient, suggesting that shepherds ignite fires to increase fodder supply for grazing livestock. Our result thus confirm observations that shepherds increase fire frequency to improve fodder for their herds (Xanthopolous 2000; Papanastasis 2004), and thus that fires may become more frequent where grazing and shepherds concentrate.

Taken together, our results support the climate and land use hypotheses (especially the link of fires and the abandonment trajectory), while the recreation hypothesis was rejected based on our results. Interestingly, we found high fire frequencies at both ends of the land use intensity gradient, i.e., where traditional land use systems are abandoned, and where land use intensification was strongest. This suggests that the inverse U-shaped relationship between land use intensity and fire frequencies in landscapes where humans are the primary cause of fires (Lavorel et al. 2007) may not hold for Crete, and questions the generality of the inverse U-shaped patterns.

Our results also showed that no single group of drivers explained fire frequency well, suggesting substantial interactions among drivers, These interactions are likely related to the fact, that fire occurrences require both, ignitions and the availability of flammable fuel. For example, rural exodus is associated with an increasing fire hazard due to the accumulation of flammable fuel, however, without the presence of ignition sources, be they natural (e.g., lighting during summer storms) or anthropogenic (e.g., intentional or accidental burning), fires will be absent. Similarly, without a minimal amount of flammable vegetation to burn, a high number of human-caused ignitions (e.g. in the case of high population density) rarely results in the development of fires larger than our minimum mapping unit of 1 ha. Likewise, shepherds will traditionally set fires only where sufficient biomass has evolved.

Our combination of best subsets regressions, hierarchical partitioning, and Bayesian Model Averaging proved to be highly useful to assess the complex relationship of fire frequency patterns and climate factors, anthropogenic activities, and land use. Our methods complemented each other well, with best subsets regressions and hierarchical partitioning revealing variable selection patterns and variable importance within model dimensionalities, and BMA providing a summary measure for the direction and influence of the variables, while at the same time avoiding overfitting. Our modeling results were robust (e.g., similar variable selection for all three approaches), yet a few sources of uncertainty need to be discussed. First, we modeled fire patterns at the community scale, the finest scale for which many of our socio-economic variables were available. However, land use decisions are taken at the level of individuals and the aggregate scale may have hindered the detection of stronger relationships between rangeland management variables and fire frequencies. Second, our study could not address the mobility of shepherds across community borders. Although shepherds usually stick to their community, we cannot fully rule out a neighborhood effect regarding ignitions. Similarly, stocking rates were calculated based on livestock numbers per community thereby ignoring that shepherds may also own rangelands in other communities, affecting grazing pressure and fuel availability there. Third, climate data were relatively coarse and derived from modeling results. Climate trends were calculated for a relatively short time period (19 years), which might be too short to reveal precipitation patterns of the past decades across the island. More fine-scale data, although presently to our knowledge not available for the entire island, may help to further disentangle the link between climate and fire occurrence. Fourth, our road map was relatively coarse and we approximated road density for the entire community. An improved estimate would be to calculate road density for rangeland areas only, as infrastructure in rangelands has increased drastically over the last decades due to by financial support from the European Union (e.g., European Rural development funds). Finally, despite our large set of predictors, we relied mainly on proxy variables and therefore cannot fully rule out missing covariates. For example, a more direct measurement of shepherding activities (e.g., # of shepherds), more detailed information on livestock husbandry systems (e.g., # animals in sheds vs. in rangelands, part-time vs. full-time livestock breeders), or spatially detailed variables regarding other human activities (e.g., # of hotel stays per community) could have further improved our models, but were unfortunately all not available to us.

Nevertheless, our results clearly underpinned that climate drivers and the ongoing rural exodus trend were the primary drivers of the fire regimes on Crete. Both drivers result in fuel accumulation as increasing summer rain leads to higher vegetation productivity and less gazing results in less biomass removed. Together, this will lead to more fuel in the landscape and a more homogeneous fuel distribution. As drier and hotter summers are also getting more frequent in the Mediterranean Basin our results suggest an increasing influence of climate on fire regimes on Crete. It is thus plausible that the fire regimes on Crete will shifts from a fuel-limited fire regime (where grazing controlled fire via biomass removal) towards a drought driven regime (where fires will be more common due to fuel-dense landscapes), a process that has began in the Western Mediterranean Basin in the 1970s (Pausas and Fernández-Munoz 2010).

Large-scale fuel accumulation due to the abandonment of traditional grazing systems and due to increasing summer precipitation will also lead to large and likely more severe fires. It is also possible that fire frequency will increase, considering that land use is concentrated in some areas, climate change leads to a higher frequency of fire-prone conditions, and ongoing urban sprawl and tourism development lead to an extension of the wildland-urban interface and subsequent ignitions. These trends are worrisome as larger and more frequent fires, as already witnessed in the Mediterranean Basin (e.g., Greece in 2007 (IPCC 2007), Portugal in 2003, (CBD 2010)) or other Mediterranean Basin (e.g. California in 2008, 2009, Australia in 2009) threaten people and their properties, degrade ecosystem services, affect biodiversity (Shakesby 2011).

Our study highlighted the complex interactions between climate, people and land use in driving fire patterns, and the risks that ongoing socio-economic developments and the transformation of land systems may bear for the future. An effective fire mitigation policy that counteracts ongoing trends towards more frequent and more severe fires, and that strengthens the resilience of Mediterranean landscapes cannot ignore these linkages. Fire mitigation policy needs to go hand in hand with rural development policy and landscape planning, to considers socio-economic trends in rural areas in order to reverse rural exodus trends, and to implement rural development policies that support traditional land systems (Moreira et al. 2011). This first and foremost requires mapping vegetation trends and understanding land system change. In the long run, fire mitigation policy and rural development policies should jointly work towards preserving traditional links between nature and society and to forge new links where traditional livelihood strategies are no longer viable (Fischer et al. 2011).

Acknowledgements

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Chapter V:
Synthesis

Land use is crucial for the survival of humanity, but involves substantial trade-offs as land use transforms natural ecosystems, triggers biodiversity loss, results in the degradation of many ecosystem services, and thus may ultimately threaten human-well being (MEA 2005c; Leadley et al. 2010). The demand for land-use-related products further increases rapidly due to population growth, shifting diets, and ambituous bioenergy policies, while the remaining land resources for meeting these demands diminish due to land degradation and forest protection policies (e.g., REDD). Humanity faces the grand challenge to produce more while substantially lowering the environmental impact of land use (Foley et al. 2011). Meeting this challenge requires a better understanding of the dynamics of land systems, both regarding *where* land use changes and *why* land use changes.

A better understanding is particularly crucial for the world's dryland areas, covering about 41% of the earth's surface, sustaining 2 billion livelihoods, harboring unique biodiversity and providing globally important ecosystem services. These ecosystems are characterized by harsh environmental conditions, including highly variable rainfall and shallow soils. Past land use has sometimes resulted in the degradation of these ecosystems, and climate change is increasingly exerting additional stress on livelihoods and land use systems in many dryland regions. Quantifying the patterns and drivers of land system change in such environments is therefore of great concern. While remote sensing is arguably the most valuable tool for mapping land use and land cover change (via mapping vegetation change), this task is challenging in dryland areas, as changes in land use frequently occur gradually (e.g., increasing grazing pressure) and thus result in land cover modifications. These subtle changes can be masked due to the large natural variability, for instance due to phenology, and due to disturbance-type events such as fires. Moreover, some dryland ecosystems, such as Mediteranean regions, are often characterized by a large heterogeneity and small-scale land use pattern, further complicating the separation of natural variability from land-used induced land cover change.

The recent opening of the Landsat archives offers unique opportunities to reconstruct fine-scale land use histories for dryland ecosystems in general, and Mediterranean ecosystems in particular, for example via trend analyses based on yearly Landsat time series. The second chapter of this dissertation addressed an important question that needs to be solved to allow for effective mapping of such ecosystems: "*What are the trade-offs between different vegetation estimates when using trend analyses to assess gradual vegetation change in Mediterranean ecosystems?"* The comparison of vegetation estimates clearly showed that the spatial patterns of vegetation change are similarly well captured by different vegetation estimates. While spectral mixture analysis outperforms other vegetation estimates for single-date applications, trends based on much less complex vegetation measures like Tasseled Cap Greenness, SAVI, NDVI differed by less than 5% from SMA-based trends. Together, the results suggest that trend analyses based on simple vegetation measures allow for effectively mapping land cover modifications in the world's drylands. As the trade-offs involved when using simpler, qualitative indices was comparatively low, the results from Chapter II open up pathways towards mapping land use and land cover change across large areas in dryland and Mediterranean ecosystems, while exploring the temporal depth of the Landsat archives. Landsat data are now freely distributed and available in preprocessed formats that allow for the establishment of consistent dense time series. Relative image processing techniques (relative radiometric normalization, relative cloud detection) are increasingly developed and entire protocols enable the processing of large data sets which involve more than one footprint. Moreover, several missions are currently underway to extend the Landsat-data record into the future, most importantly the Sentinal missions of the European Space Agency and the Landsat Data Continuity Mission of NASA. Methods such as those applied in Chapter II will therefore be of growing importance in the future for assessing land system change, and bear great potential for better understanding land use change in dryland ecosystems.

The third chapter sought to apply the methods developed in Chapter II to map land use and land cover change for a comparatively large region, the entire island of Crete, Greece. Crete, as many regions in the Mediterranean Basin has a long land use history resulting in very small-scaled, mosaic-type landscapes and Crete is fire prone due to its characteristic climate with hot and dry summers. Also typical for many Mediterranean regions, the island has experienced widespread socio-economic changes, resulting in a strong land use polarization with intensification on productive land and abandonment of marginal land and traditional pastoral practices elsewhere. In rangeland areas, both intensification and extensification tend to result in gradual vegetation change, fires are also frequent on Crete. Mapping land use change in Crete's rangelands therefore requires separating disturbance-type vegetation changes, like fires, from gradual changes due to land use change. Moreover, fire and land use are linked, as abandonment may lead to fuel accumulation and increase fire hazard, and a concentration of livestock herding may lead to increased ignitions as fires are often use by shepherds for rangeland improvement.

To answer the question *"How have different land use change processes and fires reshaped vegetation communities on Crete?"* Chapter III adapted a trajectory-based time series analyses (the LandTrendr approach, Kennedy et al. (2010)) that had so far only been applied in temperate and boreal forest ecosystems to the conditions of dryland and Mediterranean ecosystems. The analysis clearly highlighted the great potential that such methods have to map gradual and disturbance-type vegetation change in a robust manner across large areas (i.e., many Landsat footprints). A major outcome of this study was also that the LandTrendr approach was particularly powerful when using different indices in combination, for example a vegetation measure (here: Tasseled Cap Greenness) for capturing gradual vegetion change and the Normalized Burn Ratio index for mapping fire scars. Land cover changes on Crete were complex in terms of gradual vegetation changes suggesting that both abandonment and intensification of grazing systems co-occurred on Crete. A major finding was that the fire regime remained almost constant with frequent fires in the foothills areas, whereas increasing vegetation cover in the mountains suggests a reduction of grazing pressure. The abandonment of traditional grazing practices like transhumance, increasingly leads to fuel accumulation and a higher risk of large and severe fires. Finally, Chapter III also highlighted that fire patterns are likely related to both, climate factors and land use factors, but that disentagling the relative importance of these factors can be challenging.

The third chapter therefore asked "*What is the link between different land use processes and fire regimes on Crete?"*. A quantitative analysis based on three complemetary model selection routines was used to shed further light on the relative importance of climate, people, and land use for fire regimes, and to assess the importance of different land use trajectories (abandonment vs. intensification) for the islands fire regimes. The main outcomes were that climate factors and land use, in the form of the abandonment trajectory, were the primary drivers of the current fire regime, whereas the land use intensification trajectory was less important. These findings suggest that fuel accumulation will increase the fire risk in future and fire regimes on Crete likely shift from a fuel-limited fire regime in the past to a drought-driven fire regime.

The results of the third chapter also show that gradual and disturbance-type changes can be separated in a reliable way, and chapter four showed how the wealth of data generated by approaches such as LandTrendr can be used to provide new insights into land system dynamics and the complex links between climate, people, and land use in driving fire regimes.

The fourth chapter highlighted the complex interactions between different drivers of fire regimes and the importance of socio-economic transformation in rural areas. These facts need to be considered in fire mitigation policies. This requires also considering rural development policy and landscaping planning to reverse rural exodus trends as well as implementing rural development policies to support traditional land systems. A continued monitoring is therefore important and the results from this study are a major step forward towards an automated monitoring of land systems in drylands in Mediterranean ecosystems.

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Eidesstattliche Erklärung

Hiermit erkläre ich, dass ich die vorliegende Arbeit selbständig und unter Verwendung der angegebenen Literatur und Hilfsmittel angefertigt habe. Die aus fremden Quellen direkt oder indirekt übernommenen Inhalte sind als solche kenntlich gemacht.

Ich habe diese Arbeit nicht an einer anderen Universität zur Bewerbung um einen Doktorgrad eingereicht und es existiert kein laufendes Verfahren zur Promotion. Ich besitze keinen Doktorgrad.

Ich kenne die Promotionsordnung der Mathematisch-Naturwissenschaftlichen Fakultät II der Humboldt-Universität zu Berlin in der Fassung vom 17. Januar 2005, zuletzt geändert am 13.02.2006, und das sich daraus ergebende Verfahren.

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Berlin, den 31. Dezember 2011